Can Markets Protect Biodiversity?

An Evaluation of Different Financial Mechanisms

By Arild Vatn, David N. Barton, Henrik Lindhjem, Synne Movik, Irene Ring and Rui Santos

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PREFACE

This report discusses the strengths and weaknesses of increased use of market-based mechanisms in protecting biodiversity and its associated services. It has been written in response to the latest call by the Conference of the Parties to the Convention on Biodiversity (COP 10, Nagoya) where Parties were invited to submit information concerning to what extent innovative financial mechanisms could be used to support the three objectives of the convention.

The report has been funded by the Norwegian Agency for Development Co-operation (NORAD), and is the result of cooperation between researchers at the Department of International Environment and Development Studies (Noragric) at UMB and the Norwegian Institute for Nature Research (NINA). David N. Barton and Henrik Lindhjem have participated from NINA, while Synne Movik and Arild Vatn have represented Noragric. Irene Ring from the Helmholz-Centre for Environmental Research (Germany) and Rui Santos from New University of Lisbon have also participated as co-authors of parts of the report.

On behalf of the authors,

Noragric, UMB, 26.05.11
Arild Vatn

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ACRONYMS

A$	Australian dollar
BBOP	Business and Biodiversity Offsets Programme
CBD	Convention on Biological Diversity
CDM	Clean Development Mechanism
CER	Certified Emission Reduction
COP	Conference of the Parties
CRP	Conservation Reserve Program
CTF	Conservation Trust Fund
EBI	Environmental Benefit Index
EFT	Ecological Fiscal Transfer
EFTEC	Economics for the Environment Consultancy (U.K.)
EIA	Environmental Impact Assessment
ES	Ecosystem services
EU	European Union
EU FP7	EU 7th Framework Programme
FAO	Food and Agriculture Organization of the United Nations
GEF	Global Environment Facility
ICMS	*Imposto sobre Circulação de Mercadorias e Serviços*
LULUCF	Land use, land use changes and forestry
MBI	Market-based Instruments
MEA	Millennium Ecosystem Assessment
MES	Markets for Ecosystem Services
NINA	Norwegian Institute for Nature Research
NGO	Non-Governmental Organisation
OECD	Organisation for Economic Co-operation and Development
PDR	Purchase of Development Rights
PES	Payment for Ecosystem Services
PGP	Provider Gets Principle
PPP	Polluter Pays Principle
PS	Present status
REDD	Reducing Emissions from Deforestation and Forest Degradation
SEA	Strategic Environmental Assessment
TEEB	The Economics of Ecosystems and Biodiversity
TDR	Tradable Development Rights
UMB	Norwegian University of Life Sciences
UNEP	United Nations Environment Programme
UNFCCC	United Nations Framework Convention on Climate Change
USD	US dollar
VAT	Value Added Tax
WWF	World Wide Fund for Nature

SUMMARY

There is an increasing anxiety that the international community is not making sufficient effort to halt the loss of biodiversity. Currently, new ways are sought to increase the financial basis for biodiversity protection. At the 10th Conference of the Parties (COP) to the Convention on Biological Diversity (CBD) in Nagoya in 2010, it was decided to invite the Parties "… to submit information concerning innovative financial mechanisms that have potential to generate new and additional financial resources as well as possible problems that could undermine achievement of the Convention's three objectives".

This report is a response to this demand. The aim has been to examine the opportunities and limitations of the most discussed market-based instruments and financial mechanisms within the conservation debate. While the analyses are general, there is some specific emphasis on issues of relevance for developing countries. The report is divided into three parts. The first part contains a general evaluation of market-based mechanisms, their foundations and demands. The second part is oriented towards an analysis of experiences, with some key examples of present market oriented systems, mainly payments for ecosystem services (PES) and the associated system of conservation trust funds. In the last part we look at several mechanisms that may be considered more experimental – at least in a developing country context. These include PES procurement auctions, habitat banking and ecological fiscal transfers. The potential to reform subsidies is also included in the analysis.

General findings

The idea of making more extensive use of market-based mechanisms – as opposed to legal regulations or public payments – seems to partly reflect the prevalent belief that such mechanisms will be more efficient. There is also the notion that markets may be better at raising the necessary funds. The report emphasizes that establishing markets necessitates government actions. The most fundamental such action is defining rights, as the clear definition of rights is crucial for creating financial flows for biodiversity protection. Hence, public authorities have to specify if rights should rest with land-owners or with those wanting protection, i.e., whether the 'provider gets' or the 'polluter pays' principle should take precedence.

Establishing rights to facilitate market trades demands that the goods or services involved are defined and demarcated. This is particularly demanding in the case of biodiversity and ecosystem services. Such goods and services are the result of complex processes that are interlinked and difficult to demarcate. Information costs are very high. Hence, where payments are used, they are typically linked to proxies in the form of e.g., certain practices or management options, rather than the services themselves. The multidimensional nature of the values involved, and the fact that it is often not meaningful to measure or define biodiversity in monetary terms, implies further limitations to using markets as a way of ensuring biodiversity protection. Finally, ecosystem services are inherently public goods. In a market context, this creates free-rider problems, which poses severe restrictions on the way such markets may work.

The systems studied in this report can be grouped into two broad categories. First, there are payment systems, so-called 'payments for ecosystem services' (PES). Then there are various types of cap-and-trade systems. In the case of PES, it is the payment that is creating the incentive

for protecting biodiversity or ensuring sustainable use. In the case of cap-and-trade, it is the cap that plays this role. The market – the trade – is established to reduce the costs of imposing the cap. The CBD emphasizes both conservation and sustainable use, and most systems discussed in this report are compatible with both objectives to varying degrees.

In all the systems we have studied, the state plays a key role, not only in defining rights/caps, but often also as a buyer. Creating markets is demanding. In addition to the issues of rights and demarcating the services, making the deals and ensuring that services are delivered is also costly. The number of sellers and buyers are usually high, and they are often spatially distributed. This implies that transaction costs are high – often prohibiting trades. Finally, public authority may be necessary to get around the free-rider problems, which underscores the crucial role that public authorities play in terms of establishing markets. Moreover, a major part of financial mechanisms studied in this report are all based on public programs. Hence, we observe that public authorities cannot just 'leave the problem' to markets. They have to create them and provide support or establish public programs.

Criteria for evaluation

The systems studied are evaluated using a common set of criteria:

- Legitimacy of the process
- Legitimacy of outcomes:
  - Effectiveness (delivery of services; additionality; permanence; leakage issues)
  - Efficiency (cost-effectiveness; opportunity and transaction costs
  - Equity (distributional effects)

The choice of system implies a choice between different governance structures where the legitimacy of the actual processes – how various groups are involved – is often as important as the outcomes themselves. With respect to the increased use of markets, the question is as much about whether this is a legitimate way to treat environmental values, as it is about whether markets may work well from a purely functional perspective.

Experiences with payments for ecosystem services

Looking into the specificities of each system studied in this report, PES for biodiversity protection is mostly presented as a market-based solution. Milder et al. (2010) estimate the volume of PES to be about 1,460 million USD/yr. Of this about 87 per cent is used in developed countries. A somewhat surprising finding is that as much as 99 per cent of payments derive from public sources, while this percentage is 97 for developing countries. In all the cases we have studied, payments go to land-owners, implying that they are implicitly granted the right to existing practices. High transaction costs seem to be a core explanation for the extensive involvement of public authorities. Several developing countries have established Conservation Trust Funds to operate as intermediaries between 'buyers' and 'sellers', with the aim of attracting private funds. However, public funds dominate even in these cases. So while firms and individuals do engage, the level of engagement apparently remains relatively marginal.

It is notable that little information exists on the effectiveness of PES on biodiversity conservation and sustainable use. Experience from Cost Rica – a pioneering country on PES – indicates that while effects are observed locally particularly for forest regeneration, it is hard to determine any aggregate effect on avoided deforestation for the country as a whole. Moreover, equity issues are often raised in the literature, emphasizing that if development is to be a part of PES, specific actions are needed. Drawing on studies of the carbon market, we also document that the motivations of the actors involved depends on the system. Studies reveal that firms voluntarily engaged in PES were more willing to pay extra for ensuring development than firms involved in the Clean Development Mechanism (CDM), a kind of cap-and-trade system under the climate regime. This phenomenon was observed despite the fact that there is an explicit development component included in CDM.

Experiences with new and more experimental approaches

New varieties of financial mechanisms, such as PES procurement auctions, ecological fiscal transfers and habitat banking, may potentially play a greater role in the future mix of instruments used to increase financing for conservation, and potentially create more appropriate incentives.

PES procurement auctions are considered an alternative or supplement to ordinary fixed-price or bilaterally negotiated PES schemes. With the state as buyer, the main idea is to introduce competition between landowners so that their true opportunity costs are revealed. The auction can therefore help the regulator achieve environmental objectives at lower costs and ensure a higher degree of additionality. However, PES auctions are still in their infancy internationally, though USA, Australia and a few other countries have been experimenting with such auctions. The functioning of PES procurement auctions in terms of process legitimacy, effectiveness, efficiency and equity is likely to be quite case-specific, and will depend on design elements of auctions, such as the bidding rules, to what extent information is shared, how bids are evaluated and ranked, etc. As auctions are relatively more complex, they have a higher risk of failure than a fixed-price scheme. Hence, more testing is likely to be required before conservation auctions are rolled out, especially in developing country contexts

Biodiversity offsetting is based on the idea that reduction of biodiversity at one place, the development site, can be compensated by action increasing biodiversity at another, the rehabilitation site. It can be based on a regulatory approach implying a liability to compensate damages from development. Both tradable development rights (TDR) and habitat banking are market-based instruments with trading as their main feature. Governments define a development cap, such as a percentage of land declared not available for development, or a conservation objective such as ‘no net loss’ of biodiversity. In principle, trading TDR/offsets in a market can then achieve the cap/objective at lower cost. Habitat banking opens up the scope for finding trades with even greater differences in opportunity cost by allowing credits to be banked over time. Experiences are limited to a few countries, mostly high and middle income and little empirical evidence is as yet available on the cost-effectiveness as compared to traditional regulation. Conclusions regarding potential cost-effectiveness rely on a wide array of assumptions about availability of land for trading, effectiveness in monitoring, mitigation and rehabilitation actions, assessing equivalence in habitats between development and rehabilitation sites, and compensating for inequities between stakeholder in time and across locations. In other words, transaction costs are high and there are reasons to expect them to be largely borne by the public sector. It is an empirical question whether transaction costs are outweighed by the differentials in opportunity costs between development and rehabilitation sites. It is also open to conjecture whether cost savings compensate for ineffective rehabilitation actions observed in the few pilot case studies available.

Ecological fiscal transfers (EFTs) are pioneered in Brazil, Portugal and to some extent in Germany. Decisions about where conservation areas are to be sited are frequently taken at higher levels of government, even though the costs of losing those areas for other social and income-generating developments are borne by the local governments and communities. EFTs are therefore seen as a new instrument that provides incentives for local governments to support and maintain nature conservation areas within their territories, but that can also provide wider ecological benefits beyond municipal boundaries. By building on existing intergovernmental fiscal transfer schemes, transaction costs can be kept low. Transaction costs increase where EFTs aim at going beyond mere compensation, to providing conservation incentive effects through higher compensation related stricter protection measures. The funding on which fiscal transfers are based may derive from tax revenues or redistribution of international transfers of funds such as REDD+. By addressing local government land-use decisions, ecological fiscal transfers complement a policy mix of economic and regulatory instruments largely addressing private actors.

Subsidy reform is the last action discussed in this report. It is not a new mechanism as such, but is still an important component of any mix of instruments to increase potential financing for conservation and create more appropriate incentives. Reduction of environmentally harmful or other unjustified subsidies will both free up resources in government budgets and make resource use more efficient. Reform processes are in their infancy in most parts of the world and substantial progress is necessary. Subsidies are introduced and maintained for various social, environmental and economic reasons. Many of these are both valid and/or politically rational reasons, such as those underpinning many PES schemes. While renaming ‘subsidies’ as ‘payments’ may appear to increase their legitimacy, caution is necessary. Even so-called ‘green’ subsidies may not be well-targeted or cost-effective, as also observed for several PES schemes. Removing or reducing subsidies that no longer have legitimate objectives is often a painful process for the interest groups that stand to lose. Potential conflicts can be alleviated through broad stakeholder engagement, transitional assistance, and increased transparency. The current emphasis on fiscal austerity measures by many governments during the financial crisis may create a window of opportunity for subsidy reform.

Closing remarks

Market-based instruments for biodiversity protection are to a large extent experimental and several challenges are emerging. The main concern is the very legitimacy of using markets to ensure biodiversity protection in the first place. This report has documented that governments need to play a major role in creating and regulating markets. Hence, the issue is not only about ‘how much market’, but also about the role of governments in forming markets. While trading is thought to have the capacity to reduce costs and increase effectiveness, there are several uncertainties and problems involved. These concern not least aspects of quality. We observe several challenges here. Markets can capture only a subset of the values involved. They are, moreover, best at handling well demarcated and discrete assets. Biodiversity is, however, a *system* good that is not very conducive to piecemeal strategies. What is desired is not fragmented pockets of particular ecosystem services, but the overall viability of complex systems. Ensuring that the notion of substitutability characterizing trades in offsets does not adversely impact on ecosystem function and environmental qualities is vital. This report provides an overview of the way these various challenges play out in different settings. It is a political matter to assess the relative importance of the insights and arguments it contains.

INTRODUCTION

It is now widely acknowledged that biodiversity is vital for human wellbeing, and that the current trend of declining biodiversity represents a threat to human welfare. In response to the mounting anxieties about accelerating biodiversity loss, the Convention on Biological Diversity (CBD) was established in 1992 at the behest of the United Nations Environment Programme (UNEP). The Convention is governed by the Conference of the Parties (COP), which holds regular meetings to discuss issues relating to its implementation. The Convention has three main objectives, namely conservation of biological diversity, the sustainable use of its components, and the fair and equitable share of the benefits arising from the utilization of genetic resources. These objectives are ambitious and necessary, but not easily achieved.

With respect to the overarching mandate of halting biodiversity loss and the objective of conserving biodiversity, there is a growing emphasis on finding appropriate economic tools to provide the right incentives to aid conservation efforts and sustainable use of biodiversity. In his opening address to the COP 10 in Nagoya, the Executive Director of UNEP Achim Steiner emphasized that while science is a vital tool in the investigation of the root causes of biodiversity loss and to demonstrate the links between biodiversity and other issues, economics is the key to address the issue (COP 10 2010a: 10). While it has been argued for some time that the failure to account for the full economic value of ecosystems and biodiversity is a primary cause of the continued loss of diversity and degradation of resources (TEEB 2010), increasing focus is being placed on finding economic instruments that provide both incentives to and additional financial resources from private and public actors. This increasing focus stems from the belief that, by drawing on market mechanisms, more cost-efficient solutions might be found to conservation challenges, as actors are stimulated through competition to come up with new ways of safeguarding environmental assets.

Articles 20 and 21 of the Convention spell out the need for putting in place appropriate mechanisms and architectures for financing conservation efforts. Article 21 states that “There shall be a mechanism for the provision of financial resources to developing country Parties for purposes of this Convention on a grant or concessional basis (…) The mechanism shall function under the authority and guidance of, and be accountable to, the Conference of the Parties for purposes of this Convention” (United Nations 1992). A draft strategy for resource mobilization that outlined funding targets, indicators and concrete activities and initiatives, as well as implementation and monitoring arrangements, was presented in May 2008 at the 9th COP meeting. The resource mobilization strategy was taken a step further at the 10th meeting of the COP in October 2010, in Nagoya, Japan. It was decided (decision X3, point 8(c)) to invite the Parties “… to submit information concerning innovative financial mechanisms that have potential to generate new and additional financial resources as well as possible problems that could undermine achievement of the Convention’s three objectives” (COP 10 2010b).

This decision should be seen in the light of the increasing importance attached to the adoption of market-based mechanisms in natural resource management – cf. water, forests and carbon/climate change. It is important to enhance the financial basis for biodiversity protection, with an increasing emphasis on involvement of the private sector. Market-based instruments are also increasingly proposed to better reflect the value of biodiversity/ecosystem services in market prices. However, expanding the role of markets is neither an easy nor an unproblematic avenue to walk. It raises issues concerning what are appropriate ways to protect biodiversity. Moreover, many demanding institutional changes will be needed, which is a challenge not least in developing country contexts. At the same time, it is here the need for finding additional financial resources is the greatest.

The aim of this report is, therefore, to discuss the opportunities and limitations of the most discussed market-based incentives and financial mechanisms as proposed in the conservation debate. The report is divided in three. The first part is focused on a general evaluation of the opportunities and limitations related to using market-based mechanisms. This section is written by Arild Vatn, Synne Movik and David N. Barton. The second part is oriented towards an analysis of experiences with some core examples of present market oriented systems – mainly payments for ecosystem services (PES) and the accompanied system of conservation trust funds. The analyses presented here are undertaken by Arild Vatn and Synne Movik. In the third part we examine mechanisms that are more recent and experimental – at least in a developing country context, and partly even for developed countries. Mechanisms discussed include PES procurement auctions, tradable development right/habitat banking, and ecological fiscal transfers, in addition to the reforms concerning use of subsidies, as this is a precondition in many cases for introducing new instruments. This final part is written by David N. Barton, Henrik Lindhjem, Irene Ring and Rui Santos.

While the main focus of this report is on biodiversity, we will include experiences from other relevant fields like water protection and carbon mitigation when relevant. While biodiversity protection has specific characteristics, lessons can be learned from the other fields. As there are typically more examples of market-based mechanisms in especially water and carbon services, it is of special interest to understand why this is so.

Part I:

OPPORTUNITIES AND LIMITATIONS OF VARIOUS FINANCIAL MECHANISMS TO PAY FOR ECOSYSTEM SERVICES

by

Arild Vatn, Synne Movik and David N. Barton

Introducing market-based mechanisms to ensure the delivery of ecosystem services is seen as a potential solution to the great challenges humanity is facing concerning environmental deterioration. It is, however, also acknowledged that applying market-based approaches is demanding in many respects. For one thing, it is difficult to raise the necessary finances. It is also challenging to construct ways of distributing these resources such that they reach the 'right' people and create the appropriate incentives – which in turn raises issues regarding how to measure the values involved, who should pay and how to make them pay, how should money be transferred and how is it possible to ensure that payments influence end receivers in the manner desired? More fundamentally, the question concerns when market-based systems are appropriate and when not.

The aim of this part of the report is to clarify what these issues imply for creating a stronger financial basis for biodiversity protection. We start by looking at the causes of biodiversity loss. Successful policies – whether market-based or not – need to be founded on a clear understanding of what brings about biodiversity loss. Next we present a short overview of existing financial mechanisms for ecosystem services. Thereafter, a series of sections follow in which we discuss various aspects of creating markets for ecosystem services/biodiversity protection.

Before we start on our journey into the above issues, it should be noted that it is not easy to define what is and what is not a market-based mechanism. We notice that public bodies often act as 'buyers' or 'sellers' of ecosystem services. This is not least the case for payments for ecosystem services. Instead of applying a very strict definition of market-based mechanisms, we will rather apply the broader concept of financial mechanisms. The fact that it is difficult to draw a clear line concerning this issue reflects that practice is full of various mixes of public and private, of market- and command-based systems. In the conservation policy literature 'economic instruments' has been employed to encompass roughly the same set of policies as assessed in this report – also involving different combinations of instruments (Ring et al. 2011).

1. WHAT CAUSES BIODIVERSITY LOSS?

The Millennium Ecosystem Assessment (2005:2) states very emphatically that "Human actions are fundamentally, and to a significant extent irreversibly, changing the diversity of life on Earth, and most of these changes represent a loss of biodiversity. Changes in important components of biological diversity were more rapid in the past 50 years than at any time in human history. Projections and scenarios indicate that these rates will continue, or accelerate, in the future." Tentative estimates put the rate of biodiversity loss at one thousand times higher than the background and historical rate of extinction (GBO3 2010).

However, measuring biodiversity loss with accuracy is a challenge. Ecosystems are very complex, and that diversity spans different scales. Consequently, it becomes difficult to use one particular indicator – e.g. species diversity – to monitor biodiversity changes. Establishing a set of indicators necessitates consensus on what kind of criteria should guide the choice. At present, the purposes for which indicators are applied, and sometimes also the indicators themselves, differ depending on whether they are defined in the realm of ecological science or environmental policy. There exists a variety of indicators that are giving rise to a range of, mostly incompatible, monitoring systems (Feld et al. 2009), and there is thus a need for greater transparency in the definition of criteria and selection of indicators, as well as empirically testing their relevance and usefulness (Heink and Kowarik 2010). (Feld et al. 2009:1862) note that "Despite great effort to develop indicator systems over the past decade, there is still a considerable gap in the widespread use of indicators for many of the multiple components of biodiversity and ecosystem services, and a need to develop common monitoring schemes within and across habitats. Filling these gaps is a prerequisite for linking biodiversity dynamics with ecosystem service delivery and to achieving the goals of global and sub-global initiatives to halt the loss of biodiversity."

Relating to drivers of change, the distinction between direct and underlying causes of biodiversity change is often not as clear as it may appear. There are long, complex causation chains that eventually lead to a loss of biodiversity, and few cause-effect chains are linear or unidirectional. Analyses of change are further complicated through the existence of particular feedback loops that are not easily traced (EU 2009). Despite these difficulties, it is possible to outline some main trends and drivers of biodiversity change. Five main human-induced indirect drivers of biodiversity loss can be identified: demographic, economic, socio-political, cultural and religious, and scientific and technological (MEA 2005). Increasing populations have profound implications on the world's ecosystems, not least through the increased consumption of ecosystem services. Globalization and economic growth drive ecosystem change through shaping the patterns of production. Socio-politically, there are positive dimensions with respect to biodiversity conservation, through, e.g., the trend towards greater democracy and the decline of centralized authoritarian states, which opens up for more adaptive management of environments. Culture and religious beliefs, for their part, fundamentally affect people's ideas of what they consider important, which in turn have implications for conservation practices and consumer preferences. Finally, technological change could be both a positive and negative factor. It may result in new pollutants and in access to resources that were previously 'naturally' protected. It may also reduce pressures to the extent that less resources is needed per unit of output.[1]

[1] The more technical relationships included in the above are often captured in the I=PAT formula, where I is impact, P is population, A is affluence and T is technology

Flowing from such indirect drivers are the more direct drivers of ecosystem change, such as habitat transformation, overexploitation, invasive alien species, pollution, the withdrawal of water and physical modification of rivers, and climate change – cf. Figure I.1. The single most important driver of biodiversity loss over the last century is habitat transformation, which is caused by the expansion of agriculture – cropland currently covers a quarter of Earth's surface – as well as urban sprawl, transportation infrastructure and deforestation (EU 2009, MEA 2005). According to EU (2009), agriculture causes the greatest impact on a global scale, followed by infrastructure development and deforestation.

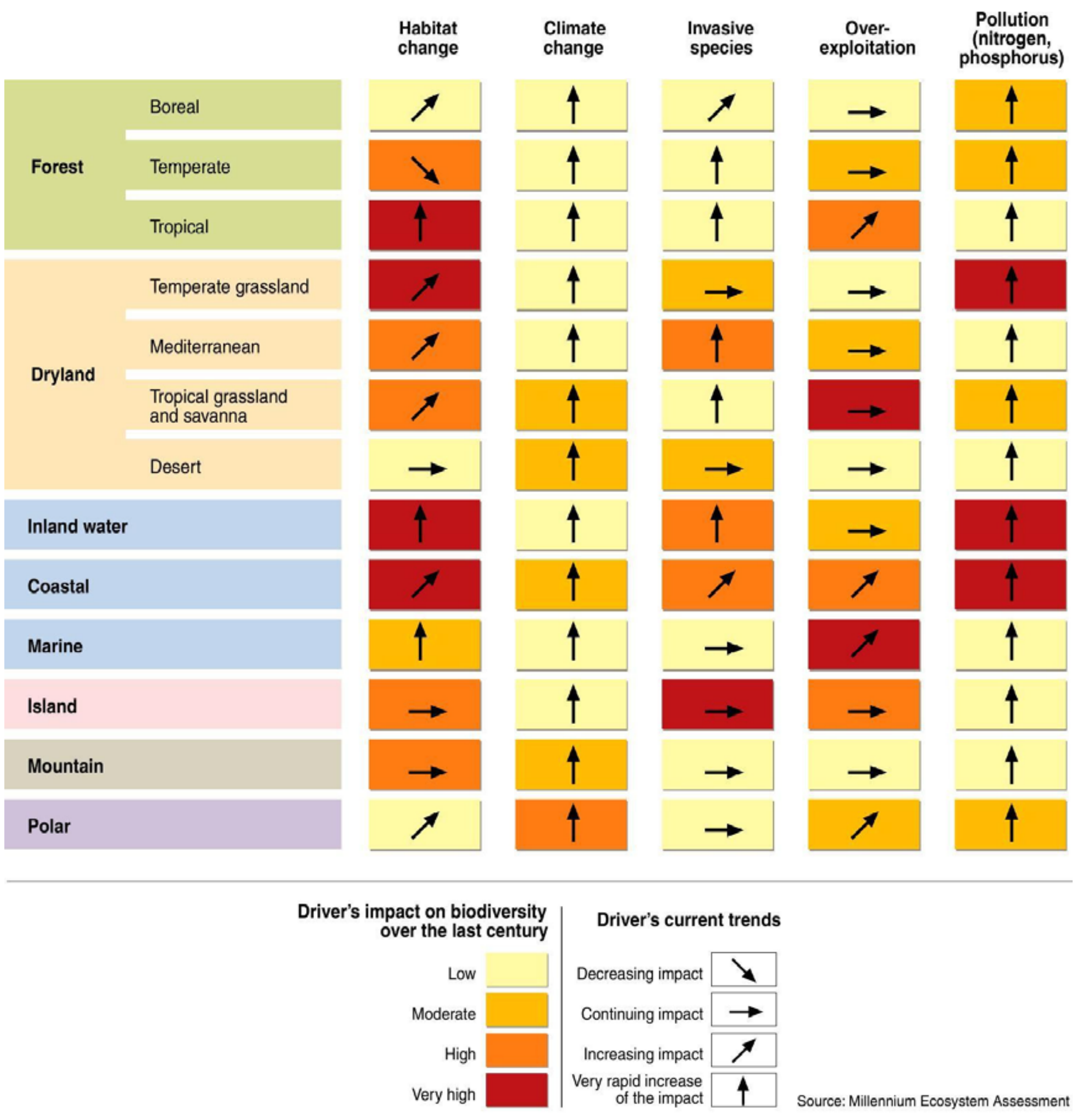

Figure I.1. Drivers of change in different biomes

Source: Millennium Ecosystem Assessment (2005)

Too intensive harvesting of resources is another significant driver, in particular affecting marine ecosystems. For example, since the emergence of the practice of industrial fishing, fish biomass has been reduced by roughly 90 per cent compared to pre-industrial levels, and three-quarters of the world's fisheries are either fully exploited or over-exploited (MEA 2005). Invasive alien species is an expanding problem, caused in part through increased trade and tourism which has the side-effect of assisting the spread of such alien species.

Pollution is another significant problem – industrial pollution accounts for more than half of the volume of water pollution, as water is used to carry diverse kinds of waste, and the most deadly pollutants. A form of pollution is the phenomenon known as nutrient loading, such as nitrogen and phosphorous, which is causing huge impacts on aquatic ecosystems. The increased use of nitrogen poses a serious threat, and is projected to become even more severe in the future, particularly in developing countries. Transforming rivers and extensive water withdrawals also play a profound role. In the 40-year-span from 1960 till the turn of the millennium, reservoir storage capacity quadrupled, and it is estimated that the amount of water stored is around three to six times the amount of water flowing in rivers.

Then there is climate change, widely assumed to become more important as a cause of change in biodiversity across biomes, through changes in temperature, increasing intensity of floods and droughts, and sea level rise. Assessing the impact of climate change on biodiversity is a complex and difficult exercise. The work on developing predictive bioclimatic models have been based on certain assumptions that are problematic, e.g. "that species distribution and assemblages are in a constant steady-state relationship with contemporary climate" (Araújo and Rahbek 2006:1396). Doing away with such assumptions makes it very difficult to predict future species distribution; different models will give rise to widely different projections, and it is difficult to fit past results with future projections. The consensus regarding possible future impacts, however, seems to be that though there may be some positive impacts from climate change, many increase vulnerability as a result of a decreasing supply of ecosystem services (Schröter et al. 2005).

Finally, consideration must be given to situations where there are interactions between two or more of the drivers of biodiversity loss; habitat loss, pollution, species migration and climate change. While the full complexity of social-ecological system may not be modeled, an awareness of the numerous interactions in any particular case where financial mechanisms are to be introduced may be a guide to the kinds of instruments that can be expected to function well.

2. TYPES OF FINANCIAL MECHANISMS

Before we start our enquiry into what it takes to create markets for ecosystem services and biodiversity, we will offer a classification of existing systems. The literature uses different schemes to categorize financial mechanisms for ecosystem services. We have chosen to distinguish between payments for ecosystem services and cap-and-trade systems.

2.1 PAYMENTS FOR ECOSYSTEM SERVICES (PES)

Many authors classify payments for ecosystem services as a market for such services, emphasizing that it is a voluntary trade between a seller and a buyer, (see e.g. Wunder 2005). In practice we observe that public bodies often act as 'buyers' and that the money involved are created on the basis of taxes or fees. In such cases the financial resources used are created using the authority of public bodies. Hence, we find it sensible to distinguish between market-based (MES) and publicly-based payment systems, depending on how the financial basis for the trade is created.

- Markets for ecosystem services (MES)

  This category includes payments where firms or individuals pay landowners – individuals, communities or states – to increase the delivery of ecosystem services. They may take the form of direct trades or trades via intermediaries, NGOs or funds and firms specializing in such transfers. From the 1990s onwards, many countries in the South established so-called Conservation Trust Funds to act as intermediaries in such trades.

  MES based contracts typically define specific actions that the 'seller' is to undertake to fulfill the contract. Certain pieces of land should be left undisturbed; particular practices are prescribed or forbidden. The contract varies depending on the aim and whether it concerns biodiversity, protection of water quality, or carbon sequestration. Karsenty (2007) refers to a specific type of MES – a conservation concession system – being parallel to ordinary forest concessions. In this case, the concession is for protection not for logging. This solution is oriented mainly towards settings in the South where forests are publicly owned, and where the state previously sold concessions to logging companies.

  Finally, we should mention certification systems where certain standards are set for the production of a product – e.g., production standards for timber. In this case, current consumers may be willing to pay a mark-up that reflects the additional costs of following the standard and undertaking the certification. In FSC certification, most of the time, there is no premium or final price differentiation upon timber sales, but it is observable that certified timber is more easily accepted by the market (Ring et al. 2011).

- Publicly based payments

  Publicly based payments take a multitude of forms. One is environmental taxes, where economic actors pay the state for the right to undertake actions that are environmentally harmful. Similarly, public authorities may pay economic actors to deliver increased amounts of ecosystem services – i.e., subsidies. Subsidies may also take the form of tax credits.

  Public subsidies are similar to the environmental payments as defined above. The core difference concerns who actually pays. Subsidies or state payments are financed by various taxes or fees. Hence, environmental taxes may be used to finance subsidies. They may also be based on standard income taxes, production taxes, consumption taxes and ecological value added taxes, etc.

A specific form of state payments or purchase is auctions. The state defines a certain type of service it wants to ensure and creates an auction to facilitate a trade over such values – e.g., Latacz-Lohmann and Schilizzi (2005). Using auctions, the idea is to create a cost-efficient delivery of these services. This system is presently rather experimental and most examples are found in the North.

Finally, a few countries – Brazil, Germany and Portugal – have used ecological fiscal transfers, and they have been suggested as new financial mechanisms for a number of other countries (Ring 2011). This is a system of conditionality where intergovernmental fiscal transfers from the state to lower level public bodies are distributed to allow the latter to provide public goods and services (e.g., concerning school systems, health care etc.). Recent indicators for distributing public revenue include the extent and partly the quality of protected areas in the specific region.

2.2 CAP-AND-TRADE BASED SYSTEMS

Recently there has been an increased emphasis on cap-and-trade based systems. It should be noted that the environmental protection in such a system lies in the cap. The trade is established as a way to reduce the costs that the cap puts on those facing it. Caps are formulated in diverse ways, giving rise to different systems. We will mention three. The first is biodiversity offsets with or without habitat banking. Here development in a certain area is only allowed if money is paid to undertake a restoration of a damaged ecosystem of similar kind elsewhere (Hartig and Drechsler 2009). In principle, the cap is hence the present status of the environment given that the offset is representing an equal enhancement to the loss created by the development. Such a "no net biodiversity loss" cap is 'global', 'status quo' and 'differential' in that no conservation target is specified for specific locations.

A second system is tradable development rights (TDRs). In this system the level of protected areas is defined as a percentage of the land. Developers may circumvent this limit by paying (other) landowners to protect more than this percentage of their land. As with habitat banking, also TDRs are mainly used in developed countries, in particular the USA. TDRs have, however, also been pioneered in Brazil (Karsenty 2007). In contrast to the above, the TDR type cap is 'location specific', 'targeted' and may be 'flat', in that it applies equally to all locations. An example of a cap is present in the Brazilian Forest Code which requires that landowners protect 20% of land in legal reserves, rising to 80% within the Legal Amazon (conversely allowing the landowner to develop 80% and 20% of the land, respectively). Both TDRs and biodiversity offsets require a clearing house mechanism which in the latter is often referred to as a habitat bank. Habitat banking may be seen as an opportunity for developers to offset their impacts.

The third type is the Clean Development Mechanism. It is part of the international climate regime – the Kyoto protocol – where countries with emission reduction commitments according to the protocol – the so-called Annex B countries – were given the right to offset some of their reduction responsibilities against paying developing countries to do the reductions instead. It is a cap-and-trade system in the sense that the Annex B countries have accepted a cap on their emissions. As the developing countries have no such cap, the system is, however, more like a PES system seen from their side.

3. PAYMENTS AND GOVERNANCE

The systems presented above are all governance systems. Historically biodiversity has mainly been protected by establishment of reserves and national parks. While this in some cases may have happened in return for financial compensation – mostly in developed countries – it was based on state command. Using market-based financial mechanisms in biodiversity protection represents a shift in the governance of biodiversity. It implies changes in who are involved in biodiversity protection and in what capacities. This concerns who formulates the goals, how decisions are made, how protection is undertaken and who has to carry the costs. These are all core governance issues.

To look more systematically into this, we start by defining the concept of a governance structure. It comprises actors and institutions, with the following three elements as core:

a) The actors involved – both those with the competence to define common goals and rules (political actors) and those with the right to use the resources given these rules in production and consumption (economic actors)
b) The institutions defining the rules for the political process
c) The institutions defining the rules for the economic process – i.e., rules concerning (i) access to and use of resources and (ii) institutions facilitating the interaction between actors

Political actors do not refer only to governments and parliaments. While ultimate power rests with these bodies, the governance literature also emphasizes the role of communities, businesses and NGOs as part of a wider set of actors engaged in the political process (Lemos and Agrawal 2006). Political actors explicitly formulate common goals for a society. Moving towards markets for ecosystem services, the ‘setting of goals’ is, however, done by market actors and in an implicit manner. It is the willingness to pay among individual ‘buyers’ that defines the level and form of protection resulting from their interaction with ‘sellers’ in a market. Nevertheless, political actors are very important even in this case as they have to define the rights which form the basis from which economic actors make their trade. The rules set for the political process (b) – who can decide about what – are a core aspect of any governance structure.

Rules defining access to resources (c(i)) typically take the form of property rights (e.g., private, state and common property), while institutions for interaction between property holders (c(ii)) could take the form either of market trade, state command or community/network interaction. Problems like loss of biodiversity may follow from the fact that rules are absent or too weak concerning the side-effects of economic activities. Hence, while rights may be defined about access to environmental resources like land, there may be no rules established regarding the use of the common or public goods provided by the land and its ecosystems.

The choice of strategy for protecting biodiversity is fundamentally about defining rights. It is about what rights property owners should have concerning the use of the resources – e.g., the land – they own or have use rights to. It is about what rights those who depend on the public goods aspects linked to that land have regarding the protection of their interests. As any use will change the capacities and value of an environmental resource, the set of rules defines who is free to shift costs upon whom.[2]

So which rights structure is the better or *most legitimate*? In evaluating the legitimacy of any institutional structure, one may typically distinguish between formal aspects and content. According to the former, a specific institution is seen as legitimate if it is chosen based on a legally accepted process. In other words it is *constitutional*. In terms of content, legitimacy relates to a general standard or ideal about what are right or just processes and outcomes. It is legitimacy of outcomes that is of interest here.[3]

In the case of environmental resources a dominant ‘ideal standard’ has been the ‘polluter pays principle’ (PPP). This principle emphasizes that those causing an environmental problem should pay. Human action may, however, also result in increased environmental quality. In relation to that the ‘provider gets principle’ (PGP) has been formulated. In practice these principles are often confused. The following simple figure may illustrate some of the challenges involved in distinguishing between PPP and PGP.

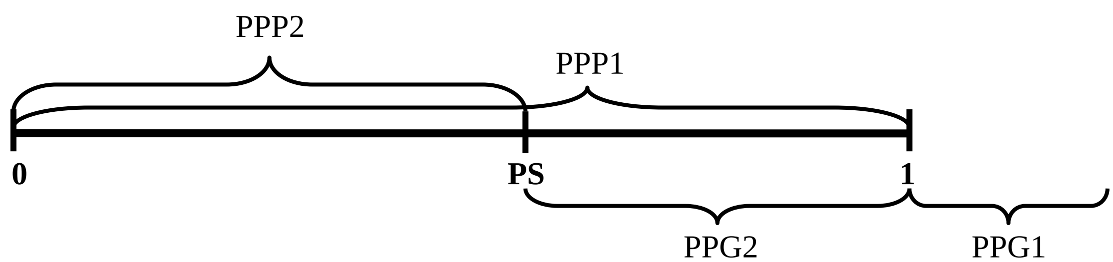

Figure I.2. Rights, compensation principles and environmental status

The line from ‘0-1’ defines the interval between a ‘destroyed’ environmental resource/ecosystem (‘0’) and an ‘intact’ or ‘undisturbed’ (‘1’) environment. PS represents the present status indicating some deterioration in the original qualities due to some types of use. This reduced quality is evaluated as positive for the resource users that have moved the system from ‘1’ to PS. They do, however, not consider the effect on others of such a change, as we assume that there are no incentives for them to do so, or no norms restricting use.

So, what are the principles or rights that could apply here? One solution could be that the one who is deteriorating the natural values should be responsible for all damages. This implies that no damage is allowed or that deterioration is accepted, but only against a payment that is set in relation to the damages caused. This rule is illustrated by PPP1. The ‘polluter’ is responsible for all damages happening in the interval ‘0-1’.

[2] Note that this goes both ways. If no restrictions are set for use of an environmental resource, costs in the form of negative side-effects will be shifted to those vulnerable to these side-effects. Conversely, if use is restricted, costs are shifted to the land owner.

[3] The literature on legitimate political action is vast. Certainly the work of Weber, Rawls and Habermas is core. Concerning legitimacy and justice in the field of the environment, we refer here to Bernstein (2001), Ikeme (2003) and Bäckstrand (2003).

Another rule is that the 'polluter' has the right to the PS (or in principle any other status between 0-1). If the environment is further, deteriorated, she must pay (i.e., PPP2). If she manages to increase the status, she will be compensated (PGP2). Similarly, in accordance with the PPP1 rule, she will be compensated only if she manages to increase the environmental quality beyond what nature itself has produced ('1') (i.e., PPG1). The latter could be the creation of a cultural landscape or building a dam that reduces disastrous floods. Certainly, such constructions may not only be considered good. It is beyond the aim here to discuss if or when such changes represent net improvements. We only assume that a development of this kind is possible.

Despite the above general principles, the decision over rights cannot be made independent of the cultural or political context in which these decisions appear. Distributional aspects and issues of *fairness/equity*, become typically very important when rules of the above kind are formulated. A simple example from REDD[4] may illustrate this. The idea behind REDD is that the North pays the South to reduce deforestation. This is seen as a cost-efficient way to reduce emissions of CO_2, hence a low cost strategy for the North to respond to its expected post-Kyoto obligations concerning emission cuts. If the PPP1 rule was instituted in this case, the South would have to pay itself for the deforestation. Given that it is countries in the North that have advocated this solution not least to lower own costs for reducing climate gas emissions (offset solution) compensation (PGP2) seems to be the more legitimate solution. Note also the added argument that the North already has cut its forests substantially as part of its development strategy and that many communities in the South depend on forests for their livelihoods.[5]

What then if the deforestation is produced by multi-national companies with owners in the North? Should these be compensated for lost access to forests in the South by asking somebody to pay them? In this case compensation (PGP2) would maybe not be as obvious. Certainly, one may again evaluate this differently if the companies have legal contracts which may make them entitled to compensation or if the 'contract' is illegal. There are many issues of this kind related to defining rules. A further discussion of relevant issues is found in Text boxes I.1 and I.2.

Legitimacy does not only concern the definition of rights. It also concerns issues like accountability, transparency, and distribution of power. Governance structures 'score' differently on these dimensions. If emphasis is on accountability and democratic decision making, more emphasis will be put on political processes and less on the market. If political processes are corrupted, this conclusion may be reversed.

From this we see that what can be considered a legitimate set of rules depends on the wider social and political contexts. No single solution can be proposed that works across all settings. At the same time, not recognizing that the above issues are at the core of choosing governance and hence payment structures is erroneous. As will be clarified later, these questions are typically not very visible in the present debate over 'new' or 'innovative' financial mechanisms.

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[4] Reduced Emissions from Deforestation and forest Degradation

[5] Note also that there is no way states in the North could obtain REDD in the South without paying/offering some kind of compensation. They have no power to define PPP1 as the rule for a country in the South. This is certainly a different story within a country as it has the legitimate power to establish whether PPP or PGP should rule. While this issue is important, the point in the text is about what can be defended as a good principle and not about who has the power to do what.

TEXT BOX I.1. Setting the standard – defining the right

As mentioned in the text, PS could in principle refer to any level between 0 and 1 in Figure 1. While in practice, PS as present status seems to play a central role in defining where PPP stops and PGP starts, there are at least two 'deviations' that need to be mentioned. Standards may be set above PS, meaning that the land owner has the duty to keep the land – typically forests – at a certain status. Such duties are often legally defined, but we also find them as parts of in certification systems.

Next, ecosystems qualities are no fixed entity. Changes in land use (PS) over time, and uncertainty about what a reference status of biodiversity constitutes (1), are principle problems in determining rights. Information about PS and 1, and technologies to mitigate pollution or provide services, also modify rights that may be legally well established, with consequences for the effectiveness of instruments. For example, even when a clearly defined biodiversity conservation target is politically accepted (for example the 'no net loss' principle implied by the EUs 2020 target of 'halting biodiversity loss') and polluter pays principles established (for example the 'mitigation hierarchy' of the EU Environmental Liabilities directive), lacking knowledge about the baseline environment before pollution, and lacking mitigation options after pollution, mean that a polluter pays principle (PPP2) moves below PS. For example, the effectiveness of biodiversity-offsets in a habitat banking scheme depends not only on its legally established 'cap', but also on the extent to which the 'mitigation hierarchy' is technically feasible and how much 'residual biodiversity impact' relative to a reference level remains to be traded with an off-site location, after all mitigation options have been tried on-site.

As a broad hypothesis, the greater uncertainty about reference levels and options, the more PPP gives way to PGP in conservation policy-making.

There are two more important issues concerning the choice of governance structures for biodiversity protection. These are more technical and concern the effectiveness and efficiency of various solutions. *Effectiveness* concerns the capacity of the structure to deliver a reduction in the loss of biodiversity. There are several issues to consider. First, one needs to evaluate how well the governance structure fits the type of good or service involved. When are e.g., markets capable of handling the necessary information well and when are they not? When can a service be easily demarcated, priced and treated as a commodity? Second, what is the capacity of the governance structure to raise the necessary resources? Third, how does the governance structure set up motivate actors to deliver increased protection of biodiversity? As part of this, how well is the structure at ensuring additionality and permanence? Biodiversity loss has certainly a long time horizon and taking action where action would anyway happen should not have priority.

Efficiency concerns the ability to deliver cost-efficient biodiversity protection. This involves both the direct cost of e.g., reduced deforestation and the transaction costs related to the chosen governance structure. Different governance systems have different ability to find cost-effective solutions. Transaction costs are also different if we compare the taxing power of a state with a more market oriented solution.

In the following we will discuss the above issues in more detail. We will start with looking at how to define the service, its value and boundaries.

TEXT BOX I.2. Rights in landscapes

Figure 1 illustrates the general rights structures that can be applied in relation to environmental services. In actual landscapes, land qualities will range from the almost completely disturbed (urban) (0 in Figure 1) to the undisturbed (wilderness) (1 in Figure 1). Moreover, we have varying information about and mitigation options for these locations. Therefore, countries have, through trial and error, adapted their policy mix to polluter pays and provider gets principles simultaneously, but differentially across different land use types. Figure 2 suggests how both PPP and PGP can exist side by side, through complementary conservation instruments because of heterogeneity in landscape characteristics/biodiversity and economic land uses/opportunity costs. In the stylised figure which is inspired by forest and environmental liability legislation in Costa Rica, a blanket ban on land use change is in force for all forests, at the same time as public protected areas form the backbone of the country's conservation strategy. Landowners are environmentally liable for damage caused to forests (PPP), while in the same landscape different PES mechanisms target different combinations of land uses with different opportunity costs (PGP). Compensation is paid for areas expropriated for national parks while PES targets land in buffer zones that is either less biologically unique and/or more costly to expropriate (both due to opportunity costs and in terms of political legitimacy).

Spatial differentiation of market-based and regulation based conservation mechanisms is costly to design and apply 'from scratch'. Costs of information about reference levels, variation in land use characteristics, and conservation effectiveness, mean that policy instruments are often developed incrementally and experimentally over time.

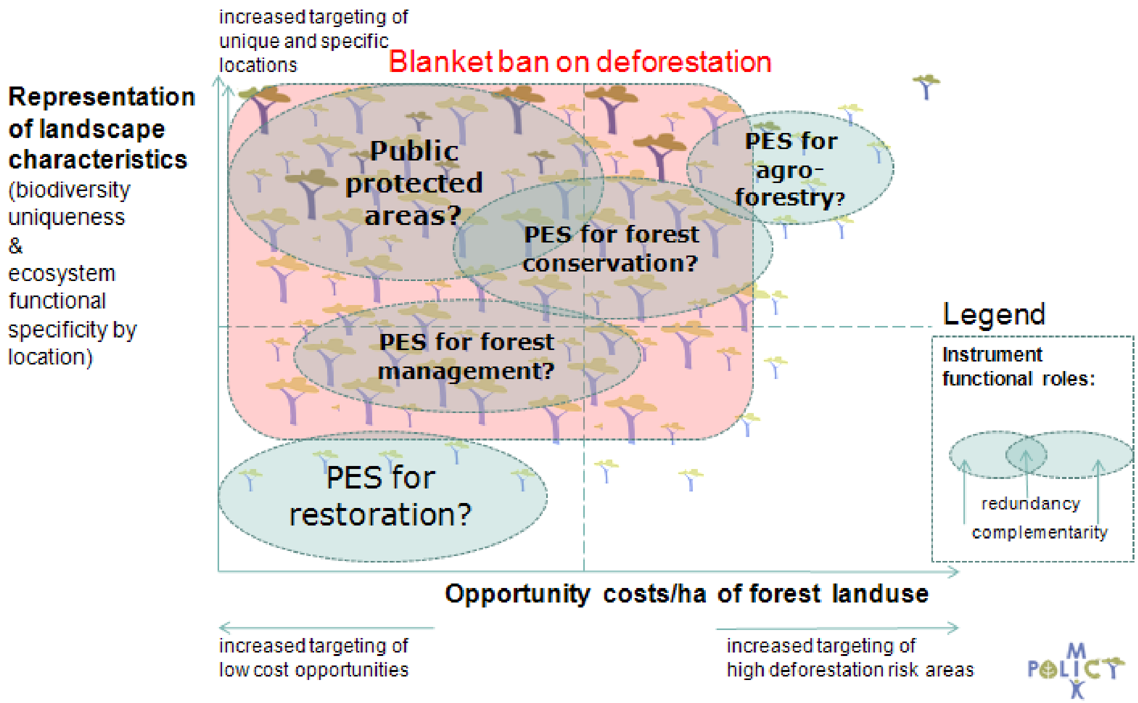

Figure 2. A 'Policyscape' for biodiversity conservation instruments.

A conceptual illustration of a PES scheme (provider gets principle) complementing a blanket ban on deforestation and public protected areas (polluter pays principle) within a landscape. A mix of principles and instruments is needed to tackle the variation of biodiversity characteristics of the landscape and opportunity costs of conservation land use (symbolised by the variation in tree type and cover across state space in the figure).

4. DEFINING AND VALUING SERVICES FROM BIODIVERSITY

As already emphasized in Section 1, biodiversity exhibits a high level of complexity. This is also captured in the definition included in the Convention on Biological Diversity, which states in Article 2: "'Biological diversity' means the variability among living organisms from all sources including, inter alia, terrestrial, marine and other aquatic ecosystems and the ecological complexes of which they are part; this includes diversity within species, between species and of ecosystems" (UN 1992).

Historically, we observe a move from seeing biodiversity foremost as variations in species richness to see it more as a systems feature. Concerning the latter, it is the structural and functional relationships between species that are emphasized, seeing species as integrated in webs of matter and energy cycles reproducing the systems. This development is also found when looking at history of biodiversity protection with a shift from species to ecosystems (Meffe and Carol 1997).

The concept of ecosystem services is developed from this latter perspective. This notion has gained momentum in the last 10-15 years as part of an extended argument for why biodiversity should be protected (Daily 1997; MEA 2005). The Millennium Ecosystem Assessment distinguishes between four sets of services:

- Provisioning (e.g., food, fresh water, wood and fibers, fuel)
- Regulating (e.g., climate regulation, flood regulation, disease regulation, water purification)
- Cultural (e.g., aesthetic, spiritual, educational, recreational)
- Supporting (e.g., nutrient cycling, soil formation, primary production)

While the above services have a strong physical and chemical dimension, it is the biological dynamics that give them their distinctiveness and provides for much of their functioning.

Introducing the concept of a service has also been part of a move towards seeing biodiversity as a product whose qualities have great implications for human living conditions. In relation to this we note that the Convention emphasizes both conservation and sustainable use. Actually, there is a continuum from full protection via managed systems to sustainable use. Ensuring human livelihoods often implies that the latter is of great importance. In this report we make no emphasis on the various types of protection as the general principles discussed in this report apply to all types. Text Box I.3 offers a brief discussion of this.

The service concept has influenced the way we think about environmental protection. A service is something we are used to pay for. If there is too little biodiversity it is because we do not pay (enough) for the services involved (Daily 1997; Daily et al. 2000). Certainly, from the discussions in Section 3 above, one could equally claim that it happens because those destroying them do not pay for the associated costs. The reports from the Millennium Ecosystem Assessment establish a conceptual model where human needs are satisfied by ecosystem services which are derived from ecosystem function which in turn are dependent on biodiversity. TEEB (2010) extends this cascade model to distinguish between benefits derived from ecosystem services that meet human needs, and the (monetary) value of those benefits.

TEXT BOX I.3. Biodiversity protection and sustainable use

The CBD rests on three pillars, namely biodiversity protection, sustainable use, and the equitable access and sharing of benefits from genetic resources. There is no principal difference between the mechanisms discussed in this report concerning their applicability to biodiversity protection as compared to sustainable use. For instance, payment for environmental services could reasonably accommodate the opportunity of existing rights-holders of continued non-consumptive and consumptive use, as long as that use remains within the limits of what is considered a sustainable threshold. However, as the section on effectiveness of environmental services (cf. part II) demonstrates, it is intrinsically difficult to determine the effectiveness of PES schemes in terms of the nature of ecosystem service delivery, and thus also to determine the relationship of sustainable use and service quality. It might not be feasible, or even advisable, to try to categorize the different payment mechanisms according to whether they are supposed to serve the goal of environmental protection, or sustainable use. Generically, one could argue that the transaction cost related to protection would be less than the costs of sustainable use, as the latter would necessarily involve some form of long-term monitoring. What is most appropriate is a ‘policy mix’ that is geared to the particular contexts of each setting.

The CBD guidelines on sustainable use – the Addis Ababa guidelines (2004) – state that what is perceived to be sustainable will vary according to the type of biodiversity, the conditions, and the institutional and cultural context in which use takes place. The underlying premise of ‘sustainable use’ is that it is possible to use ‘biodiversity components in a manner in which ecological processes, species, and genetic variability remain above thresholds needed for long-term viability’ (ibid: 2). Using mechanisms to encourage sustainable use underscores the need to develop indicator frameworks that are comprehensive, and needs to deal with issues of fluctuation and multiple equilibria (cf. Appendix). Note also that many programs geared at protection involve management – implying similar needs. Similarly, offset programs demand monitoring both of the on-site and off-site areas.

Sustainability is a very contested term, and there is a huge literature on the subject which deals with what ‘sustainability’ – hence, sustainable use – means in terms of natural resource management and ecosystem governance more generally (see e.g. Leach, Bloom et al. 2007). However, the scope of this report does not allow a detailed discussion of the concept. The guidelines also highlight the need to recognize local stewardship rights, arguing that when people’s rights of access are secure, the incentive to over-exploit resources will be removed

It should be noted that this shift in perspectives has also spurred negative reactions. MEA has been criticized for its simplistic view of the correlation between biodiversity and ecosystem services, citing a body of research which shows the relationship to be ambiguous in a number of particular, but not trivial cases (Naeem *et al.*, 2009). Sharman (2010) notes that by emphasizing the service aspect, a change in focus towards a utilitarian mode of thinking is made, and this comes at “the detriment of the idea that humans have a responsibility for nature irrespective of its notional value to humans” (p. 2). Hence, some have now started to talk about biodiversity *as opposed to* ecosystem services.[6] Spash (2009; 2011) sees the development as driven not least by a belief among conservationists that emphasizing the service dimension and the implicit mone-

[6] This was, as an example, very visible at the ALTER-Net Conference on Ecosystem Services and Biodiversity, held in Vienna, November 2010

tary value will increase the power of the argument for biodiversity protection as 'money speaks strongly'. He sees it as a pragmatic move. Both Sharman and Spash emphasize that it is the strength of the interests and developments going against biodiversity protection that causes the problems, rather than the form of the argument.

TEXT BOX I.4. Ecosystem services in Norwegian biodiversity protection

Reticence to basing conservation policy objectives solely on the concept of ecosystem services can be seen in new legislation such as Norway's 2009 Nature Diversity Act, which establishes objectives and means for Norway's conservation policy (Barton et al. 2011). A government White Paper (Miljøverndepartementet 2009) discusses comments to the hearing process from stakeholders who called for the inclusion of ecosystem services as an objective of the Act. The White Paper justified not including provision of ecosystem services among the objectives amongst others because "which ecosystem services [are important] will vary according to the specific quality and type of nature and humans' needs" ; that it would be "more clarifying [than cite ecosystem services] to mention the dependency humans have on nature as it constitutes the basis for activity, culture, health and wellbeing"; and that "if biological, landscape and geological diversity and ecological processes are maintained, then nature will supply ecosystem services to humans"(translation by the authors). The Act has among its first objectives to conserve these types of diversity. The White Paper states, however, that ecosystem services can be "a useful concept in interpreting the law and promoting awareness about the values of biodiversity for humans, and who provides and benefits from services" (translation by the authors)

Framing the issue in service terms has pushed the question of what they are worth in monetary terms. Looking into this, we note that it may not be necessary to do monetary valuation even though one intends to use payments. One can equally well define a level of protection based on other criteria. In the case of market solutions, however, monetary valuations will be important. Those buying the service(s) must make up their mind about what it is worth in monetary terms – i.e., what they are willing to pay.

There is substantial disagreement in the literature about monetary valuation of environmental amenities or services. This concerns a series of issues like whether the different values related to environmental goods and services can be sensibly measured by one common scale like money. Authors emphasize the plurality of value dimensions that cannot easily be reduced to one dimension – e.g., ethical issues and aspects related to nature and peoples' identity (cf. the arguments by Sharman above). Different types of information problems are also core examples of problems related to monetary valuation. Given the complexity of biodiversity, one may question whether individuals asked to pay or offer a price actually have the necessary knowledge to make an informed choice. Using markets to value the good may imply putting trust in consumers having very little insights in what the factual issues are. Questions are also raised concerning the problem of delimiting the good or service – to make it a tradable item. As we have seen, biodiversity is a complex system feature. As such it is very difficult to define and demarcate what the 'commodity' to be paid for really is.[7] Hence, the service could be whatever the individual thinks it could be. Spash (2011) notes that they then do not have much content. For a more in depth analysis of these issues, see Daily et al. (2000); O'Neill (2008); Vatn and Bromley (1994); Vatn (2009).

As noted, when using markets for ecosystem services, monetary valuation will form the basis for the demand. Hence, the above problems will all be relevant. When issuing state taxes or publicly defined payments ('subsidies'), the goals defined concerning reductions in biodiversity losses or delivery of specific services need not to be established on the basis of monetary valuations. Political bodies – if they demand decision support – can make the decisions based on inputs from a variety of methods or procedures – e.g., monetary assessments, expert advice/environmental impact assessments, multi-criteria analyses, deliberative methods (see Munda (2007); Vatn (2005; 2009); Wittmer et al. (2006) for discussions of these methods).

A challenge for all these methods concerns the problems of identification of biodiversity and the cognitive burdens of evaluating complex alternatives. Deliberative methods like citizens' juries and consensus conferences respond to these challenges by combining the competencies of lay people and experts. Lay people are considered competent on the value issues, while experts have technical expertise and factual knowledge. Through in-depth communication lay people develop proposals concerning what is the better choice or goal for e.g. biodiversity protection, while experts support this process through presenting what the knowledge frontier looks like and what seems to be the main uncertainties involved (e.g., Renn et al. 1995). Deliberative methods, do not eliminate the cognitive burdens of assessing biological diversity, but may help expose decision-making problems that come from lacking information, instead of hiding them behind a seemingly exact monetary estimate.

In relation to the above, one should also note that while monetary valuations are based on individual preferences of economic actors, deliberative methods emphasize the role of the citizen. The argument behind the economic model is 'consumer sovereignty'. Consumers should be free to choose and nobody should have the right to question the preferences underlying these choices. In the deliberative model, this is differently understood. In the case of common goods like biodiversity, one individual's preferences will by definition influence other peoples' opportunities. Hence, the sovereignty of individual preferences is challenged. Instead, it is maintained that it is the soundness of various preferences and arguments that should is important, implying that deliberation and not the summing of individual payment bids is the better process to define which ecosystem services to protect or produce.

Finally, there is also the argument that the main issue concerning ecosystem services like those flowing from biodiversity, are the future risks and uncertainties involved. It is about securing the integrity of ecosystems and hence avoiding stepping beyond certain bounds where their functioning is destroyed or shifted. When this is the case, the issue is more about defining limits such that important thresholds are not exceeded. Identifying such thresholds would demand processes of a kind that are different from that of economic valuation of incremental changes in

[7] We recognize that at the COP in Nagoya there were discussions about market-instruments and the implicated commodification of nature. This is a discussion which also is emphasized in the academic literature. The issues concern the problem of perceiving environmental goods as items that can be traded.

environmental quality. It is typically an issue for experts, although the complexity of biodiversity impact assessment may also exceed combined expert knowledge (cf. the discussion in Text Box I.3 on sustainable use).

To ensure that the desired level of biodiversity protection is realized, one also needs to know the costs of changed use. In the case of market trades the idea is that these costs will reveal themselves as part of negotiating the deal. The seller will only accept payments that cover all the costs that s/he encounters. In the case of public payments – whether taxes or subsidies – these costs have to be established by the public authority in order to define what level of taxes/subsidies is needed to realize the goals. The costs will concern 'lost opportunities' – hence, they are called opportunity costs. They include lost income from selling goods and services from ecosystems like timber, non-timber forest products, food, energy services or land for infrastructure, housing etc. Certainly, livelihoods may also produce goods that are not traded – e.g., firewood and food for own consumption. Nevertheless, the opportunity costs for the people involved can be estimated on the basis of what it costs to buy equivalent amounts of these goods in local markets. Having said this, one should note that livelihoods for people may not be just 'income'. Issues like identity may be linked to the type of consumption or production practices involved. In such situations defining the necessary payments may be demanding. In some situations payments may even be seen as unacceptable by local communities.

One issue is common to both pricing and costing. That concerns the definition of the service. To value, pay or tax demand to demarcate what is to be valued, paid for or taxed. This is demanding given the fact that the service in many instances is a complexity of processes. In the case of ordinary commodities like milk, nails or gasoline, qualities and volumes are standardized and fairly easy to measure. In the case of services such as day care, haircuts and travelling, defining them is more demanding. Again, however, different standardizing procedures have been developed to make transactions possible. Some ecosystem services come close to 'ordinary commodities' – e.g. wildlife experiences/safaris. Other services, such as the regulating and supporting services as defined by MEA (2005), are much more difficult to define and delimit. Precise demarcation and definition is either very costly or impossible. In such cases, the only way open is through defining proxies. Such proxies could be land of a certain type or specified practices for certain land use classes. While the service cannot be accurately measured, these features are observable and through research one can establish relationships between protecting certain types of land cover, demanding certain practices and the degree of biodiversity protection that follows. Certainly, using proxies opens up for various opportunities to cheat that need to be acknowledged when establishing governance systems.

5. CREATING FINANCIAL RESOURCES

Financial resources for the protection of biodiversity and the attached services are created by defining rights and responsibilities. The state can define rights to be with the polluters/providers or by the victims/beneficiaries of the (dis-)service. From that basis the state can issue taxes (polluter pays principle – PPP) or subsidies (provider gets principle – PGP) to facilitate enhancement of biodiversity. Given that rights are defined, markets can also be created where those destroying biodiversity make contracts with rights holders and pay compensation (PPP) or providers of enhanced biodiversity are themselves paid (PGP).

In the case of environmental taxes, the tax serves as an incentive to take better care of the ecosystem service. As it becomes costly to diminish environmental qualities, less damage will happen. While this is emphasized as the main *raison d'être* for environmental taxes, income from such sources could also be used to finance further environmental protection activities or compensate victims for the environmental damage caused by the activities that are taxed. Economists dominantly argue that such 'earmarking' should be avoided, as it reduces efficiency. On the other hand, there is quite strong evidence that the acceptance among people for environmental taxes increases if the money is earmarked to finance environmental action (Kallbekken and Sælen 2011). This latter observation is not unimportant as the general acceptance of increased taxes among the public is rather weak and puts some important limitations on using taxes as a way to solve environmental problems

Subsidies are based on the opposite rights structure (PGP). Here economic actors are paid by the state for delivery of ecosystem services/avoiding certain activities that are environmentally detrimental. While subsidies may be politically easier to institute than taxes – the carrot vs. stick analogy – there are issues also here. Economists point out that subsidies may create the wrong incentive and result in too many firms being attracted to a sector. Moreover, to be able to pay, the state needs to raise the necessary funding. This may happen through a variety of ways as indicated in Section 2.1 – involving again what is considered acceptable general tax levels.

Defining rights in the case of environmental harm may be demanding. As shown in Section 1, it is often difficult to define who exactly causes the loss of biodiversity. The causal chain is long. We see this is typical when the loss is due to pollution or to land fragmentation. In the first case, the causes are often mixed and sources may be far away, even in other countries or continents. Hence, it is difficult to claim that a certain economic activity is (among) the cause(s). Next, the country which experiences the losses may not have the power to define obligations for the source(s) as it lies outside of its jurisdiction. In the case of fragmentation, the effect is following from a sum of independent, often small acts where each single action has a non-measurable impact on biodiversity. Who is responsible then?

While the above arguments put restrictions on any payment system, it is especially problematic for markets, as rights need to be very clear for any trade to happen. Looking into this, we should first note that according to the so-called Coase theorem (Coase 1960), it does not matter who has the right – the polluter/developer or the victim. Assuming that the costs of transacting are zero[8], the level of environmental protection will be the same. What matters is who is willing to pay the most; whether the firm is willing to pay more for building in a wetland area, or demand higher compensation for not doing so, than the 'victims' are willing to pay.[9]

[8] This assumption is not very realistic, however. It will be discussed in Section 6.

[9] There are several issues at stake here not considered in the famous paper by Coase. One concerns the effect of the monetary transfer itself. If rights are with the local inhabitants and they are more environmentally friendly than the firm, giving the right to them will result in further environmental gain as they will to a larger extent use the compensation on environmental issues. Another concerns the huge difference observed between willingness to pay (rights with the polluter) and willingness to accept compensation (rights with the victims). According to Horowitz and

For most ecosystem services trades – not least for those emanating from biodiversity – establishing clear rights and undertaking the necessary controls for delivery clearly involves very high costs of monitoring, impact modeling and enforcement. The zero transaction cost assumption of the Coase theorem, which underpins the rationale for self-interested trade between an ecosystem service beneficiary and provider, makes the theorem of limited relevance to biodiversity conservation. The observation that many different state-promoted PES systems exists gives testimony to this argument. This is the reason why most markets are found in situations where local effects are dominating. Here causal chains are acceptably simple. We will make a more thorough analysis of this in Part II.

There is one exception to the above. Voluntary payments do not demand any prior definition of rights. In this case one party is voluntarily taking on the responsibility to pay. Individuals and firms may be willing to pay for biodiversity loss because they want to support a good or important cause without necessarily expecting any return. While individuals may do this as they find biodiversity protection very important, firms may do it as part of their corporate social responsibility strategy and/or to create a positive image that may pay-off in increased sales of their products. We will return to this in Section 7 on motivation.

In the case of biodiversity protection in the South, many countries lack financial resources to compensate local communities. As these are often dependent on land conversion for their livelihoods, forcing protection without compensation may seem unethical. Moreover, it is often Northern interests that favor protection. Hence, payments from North to South are needed both due to ethical considerations and because of sovereignty issues. Protection has to happen outside the jurisdiction of those opting for it. While this situation limits whether PPP or PGP is to be applied – actually PGP is the only relevant option – it does not change the problems related to create the necessary financing as discussed above. To the extent firms from developed countries want to establish in a developing country, the latter could have the option to demand payments to e.g., biodiversity protection as a condition for establishing. Whether this would be in the interest of that country is a different story.

6. SYSTEMS FOR FINANCIAL TRANSFERS

Paying money also demands a system for financial transfers. In the case of public taxes and subsidies, the state or local authorities need to identify who should be taxed or paid. Next, the level of payment has to be decided and money transferred. In that sense the state (or local authority) could be seen as an intermediary between tax payers and receivers. The power of the state is used to issue taxes. This implies that payments to the state are not voluntary, while payments from the state – subsidies – more typically are.

In the case of using markets, buyers and sellers need to find each other. While who is a buyer and who is a seller is defined by rights, whether there will be a trade depends on how easy it is for the parties to locate partners for a trade and agree on a price. In some cases buyers and sellers

McConnell (2002), these figures are in the order of 1:3. Due to this difference, who has the right could substantially influence the level of protection.

are few. This may be the case for localized issues and direct trades may be possible. More typically for the services of ecosystems spread across a landscape the situation will be characterized by many providers and very high numbers of potential victims/beneficiaries. Therefore trading may be quite demanding to facilitate. Transaction costs[10] are typically (very) high, and markets do not 'just pop up'.

It is also observed – especially in the South – that receivers are not easy to define. Property rights may be unclear, land held in common, and it may be necessary to set up new systems for making local people able to receive and distribute payments. Certainly, the power games this may create warrant specific attention. This is the case whether pubic systems or markets are used.

Due to the high transaction costs various intermediaries may operate between sellers and buyers to reduce these costs. If an individual or firm wants to devote resources to biodiversity protection, they will soon realize how costly it is to search for projects to support. They would rather prefer an intermediary that is specializing in financing such projects. It is, however, here that the power of states and other public bodies as 'intermediaries' is most clear. Adding a biodiversity tax to an already existing tax is technically very easy and demands few resources. Creating money in markets is more demanding.

Due to all the uncertainties and control problems involved in the case of ecosystem services provision, well functioning payment systems demand cooperative parties. Without a cooperative will, there are too many ways that a deal can be circumvented and partners be cheated. To establish such an environment, taking the wider institutional context into which payments are introduced is very important (Muradian et al. 2008). This concerns both rights to land, community organization and which local norms and rules exist concerning use of natural resources.

As previously emphasized, payments from North to South may be both necessary and important. In this case specific issues appear concerning how to pay local people or communities. They may not have clarified rights to the resources involved. Hence, before any payment can take place – whether public or through markets – a process of defining such rights is often needed. This may be quite a demanding process as we already observe in the case of establishing REDD.

7. MOTIVATIONAL ASPECTS

The final aspect of financing biodiversity protection that we will look into, concerns the motivation of involved actors. This is a field where there is quite some controversy in the literature. Standard economic theory assumes behavior to be motivated by individual utility maximization, or profit maximization in the case of firms. In the public choice literature, this understanding of behavior is even expanded to the area of politics and public administrations. Politicians as well as administrators are motivated only by what personal gains a policy represents.

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[10] Transaction costs are defined as costs of gathering information, making contracts and controlling that what is contracted is also delivered. While the concept is related to market interactions, it has also become standard to use this concept for costs of interaction in all types of governance structures – e.g., the interaction between public authorities and market actors..

In the more institutionally oriented literature, the perspective is quite different. Here there is emphasis on the existence of different types of motivations. They vary between institutional contexts (e.g., Hodgson 2007). So, not only transaction costs, but also motivations vary between governance structures. While markets are seen to emphasize own interests, action within a community of people is driven more by what is seen as appropriate. In the case of political action March and Olsen (1995) emphasize that governance takes place not least by the creation of what is appropriate through forming the identities of public officials. Certainly, ensuring that norms and rules for public officials are in place and followed is no simple task, but it is available for social construction and reconstruction.

The importance of this for the governance of biodiversity, relates to the way the choice of financial mechanisms may influence the way actors operate. Several questions are of importance. First of all, we may ask how the choice of governance structure influences the level of available financial resources for biodiversity protection. More specifically; what facilitates voluntary payments? Equally important; how do governance structures influence actors' willingness to play by the rules? Since it is very difficult to control that all the money invested is used for the intended purpose, it is important to create a situation where the aims of biodiversity protection are internalized among actors. Corrupted practices and perverted incentives are general problems haunting both public administrations and markets.

The distinction between payments as incentives and as compensation is important in relation to the above. In the case of an incentive, the issue is about varying the payment according to the level of delivery; the payment is the motivation. In the case of a compensation – as the concept is used here – it is more about what is a fair reward for acting responsibly. Hence, we may distinguish between a pure 'seller–buyer' relationship and that of a compensation where the logic is to compensate for costs related to service provision beyond what an actor could be expected to do on his or her own. In the incentive case, the focus is on payments per unit of delivery. As the latter is hard to define, it may be easy for providers to undercut the buyer by providing less than the contracted improvement. In the case of e.g., habitat banking there is typically a lot of possibilities for a flexible definition of the quality of the substitute. Again we observe the effects of the complexity of the goods involved and the space it offers both for the intermediaries and for those producing the service to choose 'simple solutions'. If their aim is to keep costs as low as possible, it is expected that this space will be utilized. If the focus is, however, on how to induce appropriate behavior with an accompanied compensation, the idea is that by internalizing an appropriate way of treating biodiversity, the level of corruption and perverse behavior will be reduced. People seem to trust voluntary organizations/NGOs more than corporate intermediaries when paying for 'good causes' – for the reason that NGOs are mostly non-profit organizations, and hence are motivated by the cause of protecting biodiversity, rather than by monetary incentives.

Spash (2011) points towards another aspect of this. In the case of e.g., habitat banking this is now being presented as a new opportunity for banks and other financial institutions to grow. According to Munden (2011) they will moreover be in a position to seize a dominant fraction of the payments. While Munden's analysis is focused on the carbon market, we envision that the challenges may be even greater in the area of biodiversity protection as biodiversity is more demanding to measure than carbon. An argument for the opposite is that markets for biodiversity off-sets will at least initially be at a state/national level

The existence of voluntary payments for ecosystem services represents a puzzle. Such payments from firms could – as already indicated – be explained by the fact that it may increase their standing among consumers. What seems at first glance to be costly for the firm may hence in the end increase profits as sales go up. Nevertheless, doesn't corporate social responsibility mean anything beyond being a more 'sophisticated' way to earn more money? And next, how do we explain that consumers are willing to buy more from firms that support environmental projects? They gain nothing themselves from doing so – the 1/n problem[11]. The existence of such acts rather supports the idea that people do not only behave in ways that maximize their own utility. This is a positive message for those that want to protect biodiversity. The negative message is that it might be demanding to make a voluntary market of any size.

We should finally note that there is a fundamental motivational problem inherent in payments for ecosystem services. Payments to stop destroying biodiversity could motivate people to (threaten to) ruin it. Hence, the method works against its own aim. This kind of perversion is already observed in the case of REDD. Certainly, countries with high levels of biodiversity could ask why they are not compensated for keeping it, while those having already ruined it are paid to reduce the speed of destruction or to restore it.

8. CONCLUSION

The basis for creating financial resources to protect biodiversity lies fundamentally in the definition of the rights. Setting these up is no simple issue due to the complexities involved both concerning the type of goods/services involved and the type of existing rights concerning use of environmental resources. There are many interests that oppose the creation of rights that favor biodiversity protection. This is so independently of whether we look at markets or at public governance structures.

Both the general belief in the superiority of markets as creating efficient allocation of resources and the increased pressure on the public tax system, have pushed in the direction of more use of markets in the area of biodiversity protection. This is observed all the way from the very creation of the concept of ecosystem services to experimentation with various market mechanisms. What has been little understood is what it takes to design markets and what core role the state has to play in that process.

Our analyses have shown that there are many obstacles to a successful use of markets for biodiversity. This concerns the kind of values involved and the limitations of markets in capturing

[11] The 1/n problem refers to the fact that the individual is not alone influencing the problem. If one person pays, it will not have any (much) effect. Success demands that all (n) or at least that most people pay. This structure makes it 'irrational' for the individual to pay.

these. It concerns the problem of delimiting the services. It concerns the level of transaction costs involved and the various motivational perversions that payment systems are vulnerable to.

Some of these problems are also relevant if we look at public governance systems and public payments. Delimiting the goods is equally difficult. Motivational perversions are also a great challenge here. Public systems have, however, wider options available concerning evaluating the values involved. They have also some advantage concerning transaction costs. Using state power to tax citizens or firms simplifies the generation of financial resources. Only rather bureaucratic or corrupt systems seem unable to deliver this advantage. Finally, public systems should have the capacity to overcome some of the piece-meal strategies that by necessity will characterize markets.

Certainly, bad public governance is often observed, and comparing 'idealized' markets and public systems may offer wrong conclusions. Hence, we will do further analyses in Part II and III based on more in-depth studies of actual experiences with PES, habitat banking etc. Before we do so, we want to emphasize one challenge that is demanding for both markets and public payment systems. This concerns the overall development path of our economies and what that implies for biodiversity protection.

Payments – whether via markets or public systems – are able to correct resource allocations at the margin. They are, however, weak when it comes to directing the economic development path. Observing that economic growth has demanded increase in the use of natural resources (Jackson 2009), the question arises: Can the growth process go on continuously, or is it necessary to also think about how economies could become less dependent on growth? What kind of changes in the governance structures would that demand? This is an issue of increasing importance for rich countries, not least to create the necessary space for poor countries to catch up. This concerns also how many risks we are willing to take. While searching for solutions in the form of payments may offer some relief, it seems only to be part of the solution.

Part II:

EXPERIENCES WITH NEW FINANCIAL MECHANISMS

AN ANALYSIS WITH MAIN FOCUS ON PAYMENTS FOR ECOSYSTEM SERVICES

by

Arild Vatn and Synne Movik

The experience with new financial mechanisms for ecosystem services/biodiversity is quite variable. Some are just experimental. Others are used in a few countries only. Payments for ecosystem services (PES) is a system for which there is now quite some experience. The aim of this part is, hence, to give an overview of the insights that have developed concerning the functioning of such a mechanism. Looking at PES, we will also include a brief analysis of so-called Conservation Trust Funds (CTFs) – a specific type of intermediary that has been established in order to facilitate PES. Finally, we will compare PES with a few experiences concerning the Clean Development Mechanism (CDM). While being a climate mitigation mechanism, we find that it can expand our insights about the effects of various types of governance structures for financing delivery of ecosystem services/protection of nature.

The three governance structures covered in this part of the report are important also because they provide experience from developing countries. The mechanisms discussed in Part III – e.g., habitat banking; contract auctions – are so far dominantly used in developing countries. Certainly, we note that also PES systems are more extensively used in developed countries. Nevertheless, they also play a significant role in the South. CTFs are to a large extent established in the South. In the case of CDM, we note that it was specifically created to operate in a developing country context.

1. DEFINING CRITERIA

Analyzing mechanisms like PES demands defining a set of criteria to be used. In the literature on the evaluation of policy instruments it has become quite standard to refer to the so-called 3E criteria – effectiveness, efficiency and equity – e.g. Angelsen (2008). While covering important aspects, they are all focused on consequences. We find it advantageous to also incorporate the wider issue of legitimacy, including the processes of decision making.

Bäckstrand (2006) distinguishes between *input and output legitimacy*.[12] Input legitimacy refers to the procedures by which decisions are made, including issues such as representation, distribution of power, accountability and transparency. Concerning output legitimacy, emphasis is on consequences. Bäckstrand focuses only on effectiveness, while we find it helpful to extend an evaluation of consequences to include equity and efficiency as well – i.e., the 3Es.

In the scholarly debate on legitimacy, much importance is attached to the related concept of justice. Before we specify the content of our criteria, we find it judicious to give a brief introduction to that debate. The literature on justice distinguishes between *procedural and distributive* justice. The former relates to the fairness of the process. The latter concept is quite parallel to that of equity. While the above distinction again seems to be dividing issues according to process and outcomes/consequences, the literature is quite complex concerning the meaning of what is a just process and a just distribution respectively. Looking first at procedural justice, Rawls (1971) makes a distinction between 'perfect' and 'pure' procedural justice.[13] A process of the former kind is legitimated by the fact that it produces the outcomes sought. Hence, emphasis is implicitly at consequences. From this perspective markets could be favored if the preferred outcome is e.g., efficiency. The concept of procedural justice is focusing purely on characteristics of the process. So, if emphasis is on democracy for its own sake, the logic follows that of 'pure procedural justice'.

Concerning distributive justice, there is a wide range of ways in which this concept is understood. Focus may be on equality in the distribution of income. Others emphasize equality in access to resources or in opportunities. Some emphasize justice meaning distribution based on needs. There is also emphasis on the least advantaged.[14] In our analysis we will concentrate on income and rights structure. We will put special emphasis on consequences for the poor.

Based on the above, we have chosen to use legitimacy as the overall criterion, distinguishing between legitimacy of the process (input legitimacy) and legitimacy of outcomes (output legitimacy). The latter includes the 3Es. Each criterion is more specifically defined below. All criteria have clear normative content. While the selection of criteria and sub-criteria is strongly normative, the analysis is itself will be descriptive. The aim is to describe the various systems and the experiences obtained with respect to the various elements of the criteria.

[12] She refers to work by Fritz Scharpf as a basis for this dichotomy.
[13] Certainly Rawls acknowledges that processes are typically not 'perfect'. Hence, he also includes categories like 'imperfect procedural justice' and 'quasi-pure procedural justice'. This issue is not of importance here.
[14] Cf. the work of Dworkin, Roemer and Sen.

Specifications of the criteria:

- *Legitimacy of the process*: Concerns how decisions are made, who participates and under what conditions. Accountability, transparency and the distribution of power and rights will be emphasized.
- *Legitimacy of outcomes:*
    - o *Effectiveness*: Concerns the capacity to deliver reduced biodiversity loss. This includes also the capacity to raise funds, how well the governance structure fits the type of good or service involved, and the capacity to ensure additionality and permanence as well as avoiding leakage. Implicit is also how the system influences motivation, including the risk of corruption.
    - o *Efficiency*: Concerns the ability to deliver cost-effective results. This involves both the direct cost of action – avoiding biodiversity loss – and the transaction costs related to the chosen governance structure.
    - o *Equity:* Concerns the distributional effects of the chosen system. We will emphasize issues concerning income, linking also to the rights aspects. Special emphasis will be on consequences for the poor.

2. EXPERIENCES WITH PAYMENTS FOR ECOSYSTEM SERVICES (PES)

Payments for ecosystem services are defined in different ways in the literature. The definition by Wunder (2005:3) is the most used. Here PES is understood as:

1. a *voluntary* transaction where
2. a *well-defined* ES (environmental service, or land use likely to secure that service)
3. is being 'bought' by a (minimum one) ES *buyer*
4. from a (minimum one) ES *provider*
5. if and only if the ES provider secures ES provision (*conditionality*)

The emphasis on voluntary transactions (point 1) implies that PES is understood as a market for ecosystem services (MES). Engel et al. (2008) argue along similar lines, seeing it as a Coasean market solution emphasizing negotiations between private parties. Looking at the various examples of PES we observe that, in practice, it may take a variety of forms with predominantly public bodies/states acting as 'buyers'. Hence, Corbera et al. (2007a: 366) state that "PES are not actual markets where ecosystem services are sold to service buyers. The commodity is ill-defined, and, in most cases, governments play an intermediary role by mobilizing resources from consumers to a government fund, which then distributes financial resources to ecosystem-service stewards at a pre-established price."

Hence, PES, as defined by Wunder (2005), seems more like a theoretical concept than reflecting what is found in practice. We will therefore in the below analysis include other payment systems than those resting on transactions mainly taking place between private actors. We see PES as a wider concept with MES as a special case, acknowledging that any payment to providers of ecosystem services (ES) can be classified as PES.

Having reviewed a substantial part of the PES literature, the most noticeable issue is actually the efforts necessary to create a functioning market. First of all, rights must be defined and the 'commodity' must be demarcated. As emphasized by Corbera et al. (2007a) this is demanding. The group of users and providers must also be specified, a difficult task as exclusion is often very demanding. This explains why in PES schemes *the intermediary* – not the sellers and buyers – is actually the dominant agent. These are typically states, NGOs or private firms specializing in broker activities. The intermediary defines the good, typically establishes the group of 'sellers' and 'buyers' and often even sets a predefined price.

Table II.1 offers estimates concerning the size of various segments of payments for biodiversity conservation. Data are taken from Milder et al. (2010). The table includes also biodiversity oriented cap-and-trade systems.[15] Actually, these are not normally understood as PES (and are in this report discussed in Part III), but they are included here for the sake of comparison.

As we see, governments and multilateral organizations (e.g., GEF and the World Bank) are by far the most important 'buyers'. Moreover, most 'public sector' programs are found in developed countries. The EU and US agri-environmental schemes figure as very important elements of this category. However, also programs in countries like China and Costa Rica are mainly public, with the government as 'buyer'. It is also notable that cap-and-trade seems to be important in terms of generating capital from the private market. The share of the private (voluntary) market is around 1 per cent globally; a bit higher in developing countries. As the emphasis is on biodiversity conservation, payments for bio-prospecting etc. are not included. Here buyers will typically be private, and as indicated by Landell-Mills et al. (2002), such payments are of some significance.

Table II.1. Estimated size of payments for biodiversity conservation

Type of payment for ecosystem services	Size of payments (million USD/yr		'Buyer'	'Seller'
	Global	Developing countries		
Public sector	1450	190	Governments, multilateral organizations	Farmers, forest land owners, other private land stewards
Private, regulated (cap-and-trade for terrestrial habitats and species)	380	unknown	Public agencies (transportation departments etc.), real estate developers	Mitigation banking companies, public agencies, NGOs, private land stewards
Private, voluntary (corporate social responsibility, 'green' branding, philanthropic	10-17	5-8	Corporations, NGOs, individuals	Private land stewards, NGOs, private companies, indigenous and community groups

Source: Based on Milder et al. (2010)

[15] These concern e.g., biodiversity offsets and tradable development rights. Trades under CDM are not included here as it falls under carbon projects.

Milder et al. (2010) also offer data for water and carbon services. Payments for watershed protection are about 10 times those of biodiversity conservation. This is in itself an indication of the low level of funds available for biodiversity protection. We moreover note that the role of the public sector is even more distinct in this case. Looking at carbon sequestration, Milder et al. only include LULUCF[16]. While the total market for carbon is large, LULUCF-related activities are small. Here, however, the private market is dominating. At the same time, a substantial fraction – around 80 per cent – are undertaken in developing countries.

2.1 PROCESS LEGITIMACY

PES covers a variety of public, private and mixed systems. Hence, the mechanism also covers a variety of processes. To simplify the analysis, we will merely distinguish between public and market-based governance systems. In both cases the main participants are buyers, sellers and very often intermediaries. From the literature on MES/PES – case studies and larger reviews – we find that land owners are always the sellers, independent of whether they deliver ‘new services’ or refrain from actions that are environmentally detrimental. They are called ‘providers’ and they seem to be implicitly and exclusively granted the right to the status quo uses as indicated in Figure II.1.

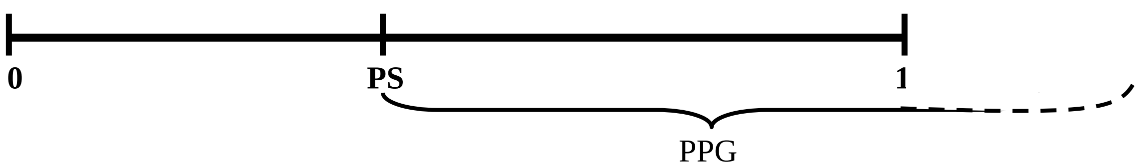

Figure II.1. Rights and compensation rules in the case of PES

We have observed almost no discussion about whether this is legitimate. Hence, the fundamental question of who should pay – what the reference point for payments should be – is hardly ever raised. This is the case whether we a) consider payments for biodiversity preservation – e.g., Wunder (2006); Claassen et al. (2008); Dobbs and Pretty (2008) – b) talk about carbon projects where developed country agencies buy sequestration from developing countries – e.g., Brown and Corbera (2003); Grieg-Gran et al. (2005); Corbera et al. (2007a and b); Wunder and Alban (2008) – c) look at local water services where downstream users pay upstream dwellers to undertake certain acts/stop certain activities to increase water quality – e.g., Grieg-Gran et al. (2005); Corbera (2007a); Kosoy et al. (2007) Wunder and Alban (2008); Muradian et al. (2008) – or d) look at combinations of all the above – e.g., Pagiola (2008).

At least from a liberal standpoint, a strong argument for PES as MES is that trades are voluntary. The users pay providers after negotiations, depending on who holds rights. We have, however, already emphasized that due to the difficulties for buyers as potential users of ES and the sellers to find each other and make direct trades, various forms of intermediaries dominate in PES. The

[16] Land use, land use changes and forestry

role of public authorities/governments is based on their power to tax citizens/apply more specific fees or levies. It also follows from the lower transaction costs (discussed later – see also Part I).

Despite its overwhelming role in PES, the legitimacy of government involvement is questioned. Pagiola et al. (2008:300) favor MES emphasizing that it "is likely to be sustainable, as it depends on the mutual self-interest of service users and providers and not on the whims of government or donor funding". This argument is very much at the core of process legitimacy. Pagiola et al. emphasize that self-interest is the right basis for biodiversity protection and that this interest is more stable than political will.

A counter-argument would not only emphasize the costs of setting up and running market systems. It would more fundamentally relate to the fact that ES are common goods, and based on that premise argue that it actually is the public and the citizens that should be the main decision-makers in defining what we want to protect and to what extent (Sagoff 1988; Vatn 2005, 2009). This is fundamentally a question about what proper issues for markets to handle are and what should belong to the public sphere (O'Neill 2007). These are issues that often get confused in the debate over PES. It is, however, crucial that in the debate over the form and extent of PES one brings this key issue to the fore.[17]

This relates also to the issue of accountability. To the extent that countries are governed democratically, political actors are accountable to the public. In markets, accountability is an issue between buyers and sellers or between the intermediary and the users and providers respectively. Moreover, being private deals, the parties control moreover the flow of information. We note that intermediaries – being governments/public bodies, NGOs or private brokers – will have a lot of power in the case of PES/MES. Hence, it is very important that their actions are transparent and not subject to various processes of secrecy. Again this relates not least to the fact that such transaction concern common goods. Certainly, governments may be corrupt and information may be concealed – but the principal difference is nevertheless important.

2.2 OUTCOME LEGITIMACY

2.2.1 Effectiveness

Turning to the 3Es, we note that the issue of effectiveness first of all concerns what results are obtained 'on the ground'. Is PES effective in protecting biodiversity? Actually, there are few rigorous empirical studies on the effects of PES on service production (Wunder et al. 2008); Pattanayak et al. 2010). Much more information exists about the way PES systems are set up, what payment formats are used, distributional issues, etc. A key reason for this is that measuring development of biodiversity is demanding. Baselines are repeatedly lacking, implying that there is no starting point against which to measure potential changes. Monitoring is also often weak or even missing outright.

As the literature does not offer a basis for an overall assessment of the effectiveness of PES, we will rather give a brief summary of a few cases that at least provide insights into the main issues. We have chosen examples from developing country contexts. One study is from Mexico. It is

[17] See also the more extensive discussion in Part I – Section 4.

actually a program for water management, but as the main strategy is to protect forests, it is of relevance to us. The other two studies are from Costa Rica; one single case and one meta-study.

The Mexican study – Alix-Garcia et al. (2010) – covers experience with the National Payments for Hydrological Services Program. One issue concerns additionality. They emphasize that the program may end up paying landowners for keeping forested land, which they would nevertheless have kept forested even in the absence of monetary incentives. The other factor that Alix-Garcia et al. highlight is the issue of leakage, where e.g. conserving forest in one area merely leads to deforestation being carried out in another area. They posit that although the mechanisms triggered by PES may work, the extent of avoided deforestation needs to be assessed at the regional or national level in international schemes. They find that the program has reduced the likelihood of deforestation by 6 to 11 per cent, which in turn represents an estimated 24 to 44 per cent decrease in the probability of any overall deforestation occurring. However, they also show that for credit-constrained households, the presence of a PES program may offer perverse incentives and trigger a process of substitution, which in turn leads to an overall net loss of forest. Such substitution effects appear to take place mostly in remote regions, supporting the notion that substitution occurs mainly where there are significant credit, land or labor rigidities. Summing up, they conclude that the program has had a modest direct effect, most notably in areas with good infrastructure, but that these direct effects may be undermined by leakage effects or through market effects induced by price changes.

The single case study form Costa Rica concerns the effectiveness of the best-known PES program in the country – the *Programa de Pagos por Servicios Ambientales* (PSA). Arriagada et al. (2011) find that there were moderate to large mean treatment effects on forest cover on farms in the study area, an impact they estimate to cover 11 to 17 per cent of the mean contracted forest area. The area of the PSA program under study, the Sarapiquí region, showed a moderate effect on forest cover, with a net increase in total forest cover on participating farms. However, it was not possible to determine whether the impact of the PSA was due to the prevention of mature forest clearance, or from forest re-growth. It was furthermore not possible to define the quality of the forest cover, or the quality of the ecosystem services provided. The last point is emphasized by Daniels et al. (2010) as well, who point out that PES in Costa Rica has forest cover serving as a proxy for ecosystem services, and explicit, site-specific indicators of ecosystem services have not been integrated into the set-up. This represents a challenge in assessing the impact of PES in terms of actually providing ecosystem services.

Arriagada (2008) finds that PES had a positive and significant impact on forest gain and net deforestation between 1997 and 2005 in the census tracts that contain at least one PES forest conservation contract signed during the first eight years of the program. Arriagada's findings are surprising considering that he only evaluated the conservation modality of PES contracts, suggesting that conservation contracts have also been allocated to non-forest lands. But a differential effect of PES conservation versus reforestation and agro-forestry modalities is to be expected

The meta-analysis (Daniels et al. 2010) covers four case studies, two national and two subregional. The study contextualizes the evolution of PES programs in Costa Rica, arguing that the starting point for PES is rather fuzzy. For instance historic forest area data are extremely variable. Another issue is to account for the roughly 178,000 ha of forest plantations in the country (FAO 2005). These factors in turn lead to a lack of clear demarcation, both spatially and over time, which has implications for measuring the real impact of PES. Nevertheless, their perhaps slightly surprising conclusion, which runs counter to the Alix-Garcia et al. (2010) assertion that effectiveness evaluations should occur on a national scale, is that the national studies conclude that PES has not lowered gross deforestation rates, *but* that the sub-national studies show evidence of additionality. The key to the seeming paradox is to separately identify the effect of PES on deforestation and forest gain by location (Arriagada, 2008). The sub-regional data suggested that forest land uses increased in relative attractiveness as compared to leaving land in a transitional state of *charral.* This brings to the fore a dilemma with national-scale analysis of PES programs that have been poorly targeted, namely that “one could erroneously determine that there were no impacts and thus undermine funding and political support for the program when in fact the program may have heterogeneous impacts (works in some places and not in others). Heterogeneity can arise because of heterogeneity in the quality of implementation or heterogeneous responses by different subgroups in the country” (Arriagada et al. 2011: 26). They therefore recommend that PES program designers should incorporate *ex ante* evaluation designs into the project implementation, to allow for better assessments and under-standing of the dynamics playing out.

Daniels et al. (2010) also note that the present legislation, which authorizes payments meant to internalize the benefits of ecosystem functioning, simultaneously bans the clearance of forests (see Text Box I.2 (Part I)). According to the authors almost 90 per cent of the PES contract areas could have been kept forested if landholders had abided by the law. However, they also state that PES may serve as a “pre-condition for the application of Article 19 since the ban on forest clearing probably would not have been politically feasible without PES” (ibid.:2119). Hence, payments can be seen as a necessary compensation for lost use rights – see also Pagiola (2008).

Turning next to the issue of fundraising capacity – which defines the aggregate size of PES projects – we note that the international community is not satisfied with the present status. This dissatisfaction has resulted in the emergence of the resource mobilization strategy being empha-sized at both CBD COP 9 and 10 – see the introduction to this report. Table 1 also indicates that funds for PES are small. This is especially the case for developing countries. In relation to that, we should also note that in the case of PES programs in both Mexico and Costa Rica more landowners have wanted to participate than the money available has allowed for. Wunder et al. (2008) document a factor of 1:3.

Concerning the type of financial sources used at present, we have already established that public money constitutes by far the main source. Text Box II.1 gives an overview over the various ways governments raise funds for biodiversity protection. While in the case of watershed protection local public funding plays quite a substantial role – through issuing an extra fee on the water bill etc. – state and federal governments play a much more important role in the case of biodiversity protection. As state taxes are typically not earmarked, it is reasonable to assume that the share of various governmental incomes that go to biodiversity protection is equal to their share in the total budget. There are some examples of earmarked funding, though. One well-known example is the use of a percentage of the gasoline tax in Costa Rica to finance their PES program (Grieg Gran et al. 2005).

Text box II.1: Examples of mechanisms to generate government funding for biodiversity protection

Ordinary income taxes are normally the main basis for government income. Further options are:

Value-added tax (VAT)
This is the most well-known form of mechanism for governments to raise revenue,, and all industrialized countries except the US have a VAT or a near equivalent. The principle of the VAT is that the government levies a tax on the value-added, i. e. the price difference between raw materials and an improved product, at every stage of the supply chain.

Production tax
A production tax is a tax upon the production of certain industries which is substituted for an *ad valorem* tax imposed under general tax laws. It is a tax in lieu of all other taxes on the leases and minerals or the equipment used in producing or in the operation of oil wells or mines.

Consumption tax
The basic idea of a consumption tax is that a tax should be put on what we consume, rather than what we earn. It is levied on commodities or services and included as part of the retail price of those commodities or services. Consumption taxes are e.g. levied on the consumption of meat, tobacco and sugar.

Ecological value added tax
The ecological value added tax is a special kind of VAT that taxes production that has a lower ecological impact lower, making such production cheaper, and conversely, taxes production that has a bigger negative ecological impact more heavily, thus rendering such production forms more expensive.

Ecological consumption tax
Based on the same premise as the 'ordinary' consumption tax, but places a heavier tax burden on the consumption of goods and services that are polluting, or have a detrimental environmental effect.

Tourism-Based Revenues
Tourism is the largest industry in the world, and tourists are often attracted to protected areas. National parks and conservation sites can generate revenues at the site, national and even international level through visitor entrance and/or user fees, concession fees, licenses and permits, tourism-based taxes, airport or country entry fees, airplane or cruise boat passenger assessments, and voluntary contributions of tourism operators and tourists.

Resource Extraction Fees
Financing conservation through revenues from fines, fees or royalties collected from forestry, energy or mining companies is a way of holding companies accountable for damage or disturbance that result directly from their operations near fragile or high biodiversity ecosystems. Resource extraction fees are usually paid as compensation to mitigate direct impacts on biodiversity (and hence are sometimes referred to as 'biodiversity offsets'.

Earmarked Taxes and Other Charges
Earmarking sources of revenue allows governments to guarantee financial resources for environmental programs through public financing tools such as taxes assessed on certain goods and services, such as e.g. gasoline, or to other mechanisms such as bonds and lotteries. These funds are then "earmarked" for specific uses by the government, such as offsetting environmental degradation.

Flight taxes
A levy on individual travelers has been suggested in order to raise money targeted towards climate adaptation projects. Such a mechanism could also be utilized for biodiversity conservation purposes, though the link is not as direct as that of air traffic pollution and climate change.

Sources; The Financial Dictionary, Morrison (2007), WWF (2007)

The challenge of raising more resources depends mainly on the political will to do so, either through direct public spending or establishing liabilities for the private sector. Certainly, there is a potential of increasing the volume of voluntary payments. Milder et al. (2010) refer to processes among industry pushing self-regulation of biodiversity impacts via fora like commodity roundtables (referring to palm oil and soybean) and criteria for international lending (referring to the Equator Principles[18]). Nevertheless, it is through public definitions of obligations and through public taxing that any sizeable sources are foreseeable. In Section 2.3 we discuss one example of expanding this basis.

A fundamental reason for highlighting the important role of governments in financing biodiversity (taxes) or facilitating the mobilization of private finance (through e.g. caps) is the fact that ecosystem services are common goods[19]. So if some pay, the good that is produced or protected becomes available to everybody[20]. Similarly, if others pay, single actors gain nothing by paying themselves (the free-rider problem). The decision by governments has the capacity to cut across this dilemma. Taken literally, this reasoning should imply that no voluntary payments appear. We have in Part I discussed why such payments may still occur. While firms may do it out of self-interest – positioning themselves in markets where social corporate responsibility have importance for customers – the act of the customers themselves must be based on motives going beyond self-interest. The core issue is the strength of this motivation and how it can be further strengthened. Per date it is rather weak – cf. Table II.1.

While voluntary markets could play an important supplementary role in financing biodiversity protection, the information problems involved also need to be mentioned. While public authorities have the capacity to set up special systems for monitoring and evaluating the status of the environment, each firm or individual will have much more restricted insights concerning what services need protection or management. Conversely, much more resources would be needed if each firm/individual were to develop the necessary overview of the needs than if this task was delegated to specialists. Key to establishing reference levels upon which to base rights and liabilities is long term consistent and spatially representative environmental monitoring, continuity of which can only be assured by public bodies (the risk of discontinuities in environmental monitoring datasets increases when private companies are replaced in competition for monitoring contracts).

In the literature there is increasing emphasis on the importance of pre-existing institutional and cultural factors for the functioning and effectiveness of PES – e.g., Mayrand and Paquin (2004); Karsenty (2007); Corbera et al. (2007a and b); Muradian et al. (2008); Clements et al. (2010). A core issue concerns how the introduced payment system fits local relations. Several aspects are important, but perhaps the most significant is the issue of property rights. Especially in developing countries these may be unclear or contested. This creates a potential obstacle in terms of implementing PES (Angelsen and Wertz-Kanounnikoff 2008).

[18] See http://www.equator-principles.com/principles.shtml

[19] By common goods we mean both public goods and common-pool resources. These resource categories have both high exclusion costs. The difference between the two concerns rivalry in consumption/use. In the case of a public good there is no such rivalry – e.g., a service can be consumed by all without reducing its quality – typically ‘existence values’. In the case of a common-pool resource use will reduce the quality. Some services related to biodiversity fall in the former and some in the latter category.

[20] This is the so-called 1/n problem. See Part I, Section 7 for a brief discussion.

In assessing this, we should first note that 86 per cent of the world's forests are owned by states (PEFK 2011). In Africa and Asia figures are close to 100 per cent. This is in itself an important fact for the implementation of PES and other biodiversity protection measures. Next, we observe that much of these forests are inhabited by communities that depend on forests for their livelihood (Vedeld et al. 2007). Their rights are, however, often unclear and insecure (Unruh 2008). Some argue that land must be privatized to make PES work well. The literature is somewhat contradictory on this issue, however. In a study of PES in Mexico Corbera and Brown (2008) conclude that a common property regime with insecure property rights may be a constraint for forest carbon project development. Certainly, lack of title increases uncertainty for buyers, and in some cases private actors have demanded formal titling to be willing to pay. In other cases people holding land in common have involved themselves in PES projects as a way to strengthen their perceived rights to the land. PES may hence result in strengthened tenure security for local people (Corbera et al. 2007b). At the same time, the latter study shows that rights need neither be individual nor fully formalized to ensure participation in PES projects. They show that PES arrangements have been established also on land held in common. Other examples point in the same direction. In the Noel Kempff PES carbon project in Bolivia on avoided deforestation, the project developers recognized informal, customary rights of local communities which are now regarded as a key factor for the success of the project (Asquit et al. 2002).

Local conditions of importance for effectiveness go beyond property rights. Corbera et al. (2007b) observe that existing organizations and their local standing play a crucial role in the process of establishing PES. This is not surprising, given the uncertainties involved and the importance of intermediaries to set up PES schemes. Similarly, when introducing PES, one needs to be very aware of local conflicts over resources. Corbera et al. (2007a) show how PES programs may actually reinforce disputes over access and control over forest resources. However, they also show how some of these problems can be countered by emphasizing trust building and participation.

2.2.2 Efficiency[21]

With respect to efficiency, we will restrict our analysis to the issue of cost-effective delivery of defined ecosystem services. Costs are of two kinds; the costs of producing the service and the costs of administering the governance structures that are needed. The latter kind of costs is usually called transaction costs.

The costs of producing ecosystem services are typically not so much about the costs of making them as about the loss of income from alternative use, though management costs are certainly involved in some cases. Often the service is produced simply by leaving the natural resources 'untouched'. Hence, costs are opportunity costs – the income forgone by the owner when unacceptable to use the resource or land for purposes that generate income. These costs may vary substantially according to location, quality of soils, type of production etc. While Stern (2007) emphasizes that opportunity costs for forests are low, Romero and Andrade (2004) argue that they are often highly underestimated. This partly concerns which alternative products are included in the analysis and what costs to include. The costs may not only accrue to land owners. Jobs in processing industries may also be lost and if alternative income opportunities are few or

[21] This section is to a large extent based on Vatn (2010)

less attractive, there will be costs also here. Emphasizing this, one should also note that opportunity costs may be overestimated in the longer run if farmers means of substituting prohibited on-farm for off-farm activities are not accounted for (Barton et al., 2009).

In the theoretical literature on PES, it is often concluded that market systems are more efficient than state-based systems. Empirically it is much harder to find a basis for reaching such a conclusion. Wunder et al. (2008:851) believe that markets or user-based systems are “much more likely to be efficient” than government financed programs. Their argument is partly based on the idea that the evaluation of the values involved is more accurate. Budget fights within governments are avoided, and payments are expected to be better targeted. As the cases they review show, the delineation between user-based and government-based very much follows distinctions between different kinds of services characterized by different exclusion cost structures. Typically user-based programs focus on a single service that is comparatively easy to demarcate. The service is simply more commodity-like. On the other hand, government programs cover less specific services. The authors’ efficiency claim, therefore, does not acknowledge the variation in transaction costs involved as markets seem to pick the ‘lower-hanging fruits’.

We should first note that the number of involved parties is important for transaction costs. If actors are few, market trades may be the least costly in transaction costs terms. As the number of agents grows, it becomes much more costly to use markets since the number of deals increases substantially. States or local public bodies such as city councils, can much easier raise the necessary funding through taxes or fees and the negotiations with providers is simplified. Certainly, the latter demands that the public body is seen as legitimate. This may not always be the case – but some private buyers or intermediaries may face similar challenges. Likewise, the cost of targeting seems underestimated in the reasoning of Wunder et al. (ibid.). A lower degree of targeting by using e.g., flat rate payments like governments often do may be more than offset by lower transaction costs. Certainly, evaluating the costs related to loss of precision in using such systems is difficult. Barton et al. (2009) point at the potential loss of representativity. It should finally be mentioned that broader, government sponsored schemes may offer opportunities for economies of scope (Vatn et al. 2002).

In relation to this, it should also be mentioned that Corbera et al. (2007b) find transaction costs to be lower if communities are involved, as opposed to individual landholders. The authors emphasize also that the level of knowledge among e.g. farmers when contracts are made at community level tend to be lower than when they are directly involved. Hence, there is again a trade-off involved. Working through collectives reduces transaction costs, but at the potential expense of lower overall knowledge dispersal.

Wunder et al. (2008) offer some insights about transaction costs for PES projects mainly in developing countries. They find costs to be in the order of 30-100 per cent compared to the final payments. Transaction costs concerns both the setting up of the governance structures and the running of them. While Wunder and Alban (2008) and Wunder et al. (2008) emphasize that the former is larger than the latter, there are reasons to believe that this depends much on the kind of governance structure. Creating a market may be very demanding while running it is less costly. Systems based on government payments may be much easier to set up as the necessary structures (at least for collecting the necessary money) may already exist.

The costs of administering the payments may next depend heavily on the type of payment scheme as illustrated by the literature on transaction costs related to environmental programs. In this case, existing data are mainly from agri-environmental programs in developed countries – e.g., Falconer and Whitby (1999); Falconer and Saunders (2002); Vatn (2002) and Rørstad et al. (2007). The findings have, however, general value.

While flat rate payments related to easily observed resources like land tend to have transaction costs at the level of a few percent of the PES payment itself, transaction costs increase, often sharply, as soon as payments are more specific. Rørstad et al. (2007) show transaction costs estimates ranging from 1-2 per cent of the payments for flat rate acreage and livestock payments with simplified control to almost 70 per cent for a program where the payment is for a very specific amenity with substantial control involved. Falconer and Saunders (2002) document a case with a wildlife enhancement scheme where transaction costs were 110 per cent of the payment. Note should be taken of the effect of the payment level on these percentages.

In trying to explain the variation, Rørstad et al. (2007) draws on Williamson (1985). Three factors stand out as important. First of all, how easy it is to observe the good or service is crucial. Using proxies like land under certain vegetation type reduces transaction costs substantially as compared to defining and paying for the service itself. The other two variables of importance are frequency and asset specificity. Increased frequency – number of repeated deals or contracts – was found to reduce per unit transaction costs while asset specificity – how complex the good or proxy is – as expected, had the opposite effect. Certainly, experience, i.e. running systems over a period of several years, would be expected to reduce transaction costs. We do not know, however, of any study that has analyzed this.

2.2.3 Equity considerations

The literature on equity issues related to PES is quite extensive. This reflects not least concerns for the poor. Several issues are involved. Let us first note that PES implies consistent payments to 'service providers'. This happen independent of whether they actually produce the service or whether they refrain from actions leading to the destruction of natural habitats. As people dependent on natural resources, especially in developing countries, are generally poor, this implies a pro-poor emphasis implicit in PES. However, rich people will also be paid, and the worry is that they are capable of 'grabbing' most of the money. Nevertheless, the basic rights implied by PES might be seen to protect the interests of those at risk of potentially losing their livelihoods as a result of conservation efforts. In the discussion of this, two issues have been raised. The first concerns whether the poor will be enrolled at all, and the second concerns whether the poor may lose access to land on which they depend. In the following analysis we will only look at PES projects in developing countries.

Examining the first question, the general picture is that the poor/smaller land holders participate less than the richer/those with more land (e.g., Zbinden and Lee 2005; Clements et al. 2010). There are two main arguments for this. The first concerns transaction costs – e.g., Grieg-Gran et al. (2005); Pagiola et al. (2008); Wunder (2008); Wunder et al. (2008). More land will be protected per transaction if contracts are made with large as opposed to small land holders. By undertaking local institutional changes, such as organizing small land-holders, this tendency can be counteracted. The second concerns the fact that many poor have no more land than what is needed for meeting basic needs. Then there is no opportunity for them to enroll. This kind of dynamic is found a lot – see e.g., Grieg-Gran and Bishop (2004); Corbera et al. (2007b); Westermann (2007); Wunder and Alban (2008) and Wunder et al (2008). Milder et al. (2010) adds to this story by emphasizing that poor people are often less used to engage in the kind of contracting needed.

On the other hand, the poor do not lose out from not being involved, either. As payments are typically directed at covering opportunity costs, one would expect them, broadly speaking, to be neutral as far as income distribution is concerned.[22] The threat of poor people losing access to land and forest resources is, therefore a much more serious issue. For example, informed by case studies in Uganda, Nakakaawa et al. (2011)[23] bring attention to the fact that though reforestation efforts have been presented as being quite successful in the Ugandan context, they do have negative livelihoods impacts for certain groups.

Similarly, Sullivan (2010) argues that significant displacement effects can arise from introducing environmental ‘value-adding’ initiatives into what she terms survival economies. The risks to people mainly relate to the prevalent lack of formal tenure arrangements, and the often complex communitarian relationships that are based on sharing and a broad knowledge of local nature, dynamic practices and knowledge systems that external initiatives may threaten to disrupt. There is, therefore, a need to be wary of the ‘win-win’ discourse that is often presented with regard to reconciling conservation measures and poverty reduction. As Benjaminsen and Svarstad (2010: 392) argue, large conservation groups, government officials, development agencies, celebrities, tourism companies, other corporations and scientists subscribe to the ‘sustainable development historic bloc’ that “present markets, commodifications and consumerism as key tools to ensure a combination of economic growth and biodiversity conservation.”[24]

One should also note that while PES may result in strengthened tenure security for the poor, they also make land more valuable and represent a further push towards loss of access for those depending on renting land. Where rights to forests are overlapping, unclear, and based on customary usage, project developers need to be particularly alert to the traditional rights of poorer and less vocal actors (Cleaver 2002).

There are also other potential indirect negative effects for the poor from PES. First of all, they may lose jobs associated with industries based on the harvesting of natural resources. Therefore Karsenty (2007) argues that a broad definition of opportunity costs is needed to ensure that PES does not put costs on the poor. He also sees a danger in that the poor become ‘low level rentiers’. There is finally the argument that as more land is protected, prices for commodities like food will increase. This will affect those among the poor that depend on buying food. Clements et al. (2010) document an example of this effect.

[22] This is not the case where forested land is left idle, by e.g., absentee landownership (Miranda et al., 2006).
[23] Drawing also on Eraker (2000) and Lang and Byakola (2006).
[24] Referring to Igoe (2010), drawing on Sklair (2001).

Some also engage in PES in the hope that it could even reduce poverty. The idea is that there could be a win-win between protection and poverty eradication as long as payments are involved. In relation to this, the literature is pretty consistent. Tradeoffs between effectiveness, efficiency and equity are clear (Mayrand and Paquin (2004). While the poor may not necessarily lose (see above) there are few mechanisms that result in their gain. Generally, poor people tend to sell cheap (Karsenty 2007). Covering just opportunity costs will not ensure development; rather it is likely to result in stagnation. Hence, to the extent that development is included as a goal in PES, specific pro-poor initiatives and safeguards are needed. This is emphasized throughout the literature – e.g., Landell-Mills and Porras (2002); Grieg-Gran et al. (2005); Wertz-Kanounnikoff (2006); Wertz-Kanounnikoff et al. (2008).

2.3 THE TOBIN TAX – A WAY TO EXPAND THE FUNDING OF PES?

As there are financial constraints to PES, one must certainly ask whether there are other ways such programs could be strengthened than those mentioned in Text box II.1. In this section we will look briefly at the Tobin tax as such an alternative. This kind of financing system is of specific relevance as it may be used to ease the financial situation for biodiversity protection in developing countries. It should be noted that while it is based on state action, it demands international agreements.

The Tobin Tax is named after the Nobel laureate James Tobin. First conceived in 1974, the Tobin tax was initially meant as a response to the increasing volatility of foreign currency markets, which were at the time making up a major part of trade transactions. By instituting a nominal excise tax on cross-border currency transactions, about 0.1 to 0.25 per cent of the volume of the transaction, Tobin figured that such a tax would discourage short-term speculation without damaging long-term investments. Such taxes would be legislated by national governments, who would gain greater sovereign control over their currencies, but would need follow-up through multilateral co-operation (e.g. to prevent the migration of the foreign exchange market to tax havens).

However, over time the Tobin tax also came to be seen as a ‘Robin Hood’ tax, a means of taxing financial speculation to create funds that could be ploughed into particular global priority projects linked to environment and development. Though the Tobin tax has been lauded by many as a potentially worthwhile instrument of regulation and redistribution, it has met with much resistance from the financial sector. It was not until after the financial crisis that interest in the Tobin tax was resuscitated and started gaining some real traction. For instance, in 2009 the then Prime Minister of the UK, Gordon Brown, and France’s President (!!) Nicolas Sarkozy suggested that a Tobin-style tax could be used to help developing countries cope with climate change, a proposition that was followed up with a EU spending commitment of €2.4bn (Traynor 2009). The idea of the Tobin tax – more soberly dubbed the Financial Transaction Tax – is an “idea whose time has come” according to a report in the Guardian (Chang and Green 2011). The level of taxation proposed is very low – a mere 0.05 per cent – but would still help in dampening the most speculative money flows. The Tobin tax is taking on several dimensions, as it were – first it was mainly intended as a device to dampen currency market fluctuations, and then increasingly as a means to raise funding for development projects (witness for instance the recent campaign to use a FTT levy to finance health projects (WHO 2010) and also increasingly becoming thought of as a ‘green’ global tax in the sense of funding climate change adaptation projects. The proposition here is to use such a tax to fund also other projects than climate change related ones. This would demand a higher tax level than those typically discussed.[25]

3. EXPERIENCES WITH CONSERVATION TRUST FUNDS (CTFS)[26]

As emphasized, intermediaries are core actors in payments for ecosystem services. Conservation Trust Funds (CTFs), also called environmental funds, are a special type of intermediary. It is of particular interest to us as such a type of funds is dominantly organized to attract financial resources to biodiversity protection; they are active in developing countries and are typically operating outside existing governmental administrations. Such funds have existed since the early 1990s. According to Spergel and Taïeb (2008) there now exist about 50 CTFs. They are mainly found in Latin America and the Caribbean region. There are, however, some CTFs in Africa and Asia, and in some countries that are part of the former Soviet Union.

CTFs are operating at national levels and many have been established by special national legislations or decrees (Spergel and Wells 2009). They can be viewed as a type of public-private partnership as the boards of these funds are constituted of a combination of representatives from civil society, business, academic organizations, donors and government officials. Non-governmental representatives are typically in majority (GEF 1998).

The literature on CTFs is generally quite positive concerning their accomplishments. It documents high overall political *legitimacy*. This is not least explained by the fact that CTFs are often established by initiatives from the hosting state. The wide representation on the boards also strengthens legitimacy. The system built for these funds ensures in general good transparency regarding the use of money. In many CTFs the board members are appointed as individuals to avoid too close ties to specific interests – however, this practice could raise issues of accountability. We have found no discussions of this issue, though. Spergel and Taïeb (2008) emphasize that the business sector is also generally positive to CTFs.

Concerning *effectiveness*, the first point to note is that CTFs are mid- to long-term engagements. Compared to single PES project initiatives, this ensures more permanence. CTFs are, however, not set up as permanent structures. They typically have defined duration periods of 20 years or less.

It seems quite clear that the establishment of CTFs was motivated by the wish to attract more resources to national environmental protection activities like national parks and other forms of biodiversity protection. According to Spergel and Wells (2009), many finance ministries initially opposed the establishment of such a structure, but were persuaded to accept the solution due to its ability to access private funds – not least at the international level. Despite this, the main

[25] Note, however, to the extent the tax is high enough to effectively regulate financial speculation, the basis for the tax is reduced. Hence, there is a limit to expanding the tax also as an income source.
[26] This section draws heavily on Vatn and Vedeld (2011)

source of CTF finances is public; e.g., the Global Environment Facility (GEF), national public donors and also the government of the host country itself. According to Spergel and Taïeb (2008) GEF and donor funding cover together almost 75 per cent of the funding for CTFs worldwide. Hence, the already noted point that public money plays a major role for PES in general also comes through here, even if the rationale of establishing CTFs was the contrary. Spergel and Taïeb emphasize, though, that corporations, non-profit organizations and foundations play an increasing role.

Despite the fact that many of the above observations are positive, the literature emphasizes that the picture is actually rather uncertain concerning the impact of CFT activities. As for PES projects in general, the focus has been more on 'process' than 'impact' monitoring – i.e., on documenting fund raising, decision-making and allocation of funds (GEF 1998; Spergel and Taïeb 2008). As in the case of PES more generally, this makes it difficult to assess the effectiveness. Impact evaluation of CTFs face the additional problem, that causality must be established between projects in specific location and funding by CTFs as distinct from other types of project funding.

Turning to *efficiency*, it has been argued that CTFs increase costs by establishing an additional management level between buyers and sellers – see Bayon et al. (1999). This kind of argument does not seem to be very well substantiated, as also discussed earlier. Rather, professional intermediaries could reduce transaction costs. How efficient CTFs are at doing that job is a bit difficult to estimate given present data. GEF (1998) documents that funds have operating costs in the range of 25-30 per cent of the total. Spergel and Taïeb (2008) record administrative costs in the order of 10-20 per cent of total annual budgets. These lower figures may as well be explained by increasing budgets over time (fixed cost issues) as by increased efficiency. It should be expected, however, that CTFs become more professional as they gain experience. The transaction cost figures presented earlier for PES (Wunder et al. 2008) were a bit higher (15-50 per cent if recalculated in accordance with the above format). This may be explained by the fact that the latter figures not only include costs for the intermediaries, but also transaction costs for the local organization and land owners running the projects. Note also that the intermediaries in the relevant PES cases covered by Wunder et al. (2008) may in many cases be CTFs.

A reason for establishing CTFs has been to reduce corruption by making the system more transparent. It has also been an argument to ensure independence from governments by increasing private donor confidence that money will be spent efficiently and avoid resources being redirected to other government uses. The aim has been to provide continuity by preventing changes in priorities following from changes in government (Starke 1995). These arguments resemble a bit that of Pagiola et al. (2008) discussed previously. We note that there is a tendency in parts of the PES literature to distrust the political process. Some of these arguments follow experiences from corrupt systems. Some seem to be more ideologically based.

In relation to this, both GEF (1998) and Spergel and Taïeb (2008) emphasize quite strongly that CTFs ought to avoid governmental majority on the board. The reason is that one in this way ensures focus on the purpose of the CTF and avoids decisions being made along political lines. This is not a straightforward argument as it seems to assume that political priorities are generally not legitimate. The argument could be turned around, emphasizing that it is a problem with CTFs that they operate outside the general political structures and processes, implying that it is donors and not the democratic processes that decide on environmental protection issues. Certainly, the argument fares differently if the political process is undemocratic or corrupted. Similarly, the short term perspective of most political systems could be an argument for lifting long term issues like environmental ones out of the standard political process. Democratic systems may themselves establish CTF type structures precisely for that reason.

Arguments in favor of CTFs also concern their capacity to avoid the rigidity of many state administrations. In line with this, Spergel and Taïeb (op.cit.) offer examples of very bureaucratic systems for management if the CTF is underlying e.g., a ministry. The potential gain that the CTFs may represent in reducing bureaucracy depends on the kind of public management system that exists in the host country.

As already emphasized, PES may not have a well defined and well organized receiver. In relation to this, Bayon et al. (1999) emphasize that effective CTFs tend to expand beyond the role as a pure financial mechanism noting that

> "They often had to play roles in building institutional capacity and private-public partnerships, developing agile management approaches, nurturing community groups becoming involved in environmental activities for the first time, and contributing to the articulation of environmental priorities and strategies" (p. 8).

The authors note, however, that many CTFs were not appropriately set up to handle such a wider set of issues. This indicates that they were unexpected.

Turning finally to *equity*, the literature notes that there has generally been a conflict between biodiversity conservation and securing local livelihoods. This concerns not least protected area management (e.g., Vedeld 2002; Hutton et al. 2005). As CTFs are primarily geared towards supporting the financing of managing such areas (GEF 1998; Spergel and Taïeb 2008), the issue is most probably challenging also for these organizations. Spergel and Wells (2009:81) point towards a specific aspect of this stating that "CTFs sometimes struggle with governments that want to use CTFs for poverty alleviation projects which are not related to conservation."

The literature also emphasizes that CTFs have donor relations and/or a management culture with quite strong conservation values, a conservation oriented competence and they tend to not engage in community based management or outreach activities. Hence, there seems, to be a lack of a local community orientation. CTFs obviously vary in strategies and competence. Nevertheless, an increased focus on training staff to carry out more outreach and collaborative type management support seems to be needed.

4. EXPERIENCE WITH THE CLEAN DEVELOPMENT MECHANISM (CDM)

The Clean Development Mechanism (CDM) is one of three so-called flexibility mechanisms of the international climate regime – the Kyoto protocol. It is a form of offset system, where countries that have reduction responsibilities according to the protocol – industrial countries and countries in transition – can undertake parts of these in developing countries against paying compensation to them. While the main focus of CDM is on carbon mitigation, there is also a sustainability clause included. Moreover forests are included in CDM in the form of afforestation and reforestation projects.

Our analyses of the CDM will be much more superficial compared to the PES analysis. The aim of this section is simply to highlight some of the experiences with CDM to enhance the understanding of mechanisms involved when paying for ecosystem services. This includes not least motivational issues.

The financial basis for CDM is created by establishing the caps on CO_2 emissions as defined by the Kyoto protocol for Annex B countries[27]. These caps have been offered to countries for free (PGP).[28] Countries may undertake reductions at home or through trade – in this case with developing countries. Figure II.2 illustrates this. The value ‘1’ implies here no emissions of climate gases.

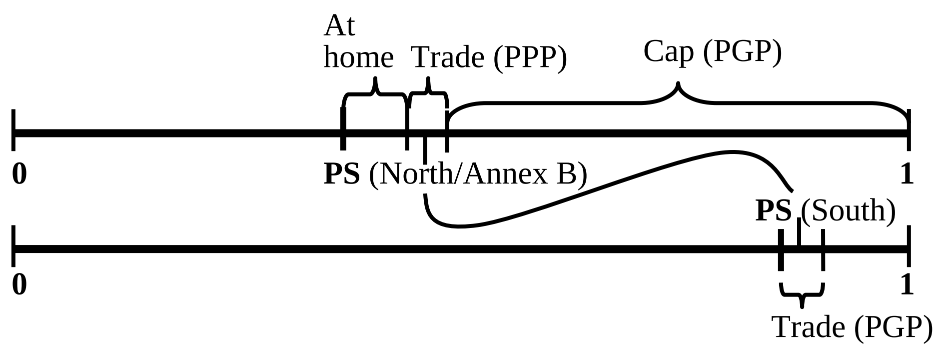

Figure II.2. Rights and compensation rules in the case of PES

While emissions are much lower in developing countries, the logic underpinning CDM is that reducing emissions there is often cheaper than to do it in the industrialized world. It is the cap that creates the environmental effects of the Kyoto regime. Hence, the *effectiveness* is fundamentally related to the level of emission cuts that are demanded by the Kyoto protocol. According to the Word Bank (2010), CDM mitigation activities accounted for 404 and 211 $MtCO_2e$ emission reductions in 2008 and 2009 respectively. This accounted for 8.4 and 2.4 per cent of the total carbon market volumes in these years. Measured in monetary terms the percentages are 4.8 and 1.8 respectively. Hence, CDM carbon is generally cheaper than other options. In evaluating these figures, it should be noted that EU – the main actor in the carbon markets – has set a rather strict limit on how much can be traded under the CDM. Of the total volumes traded under CDM, forest projects cover, however, only about 0.6 per cent of the total. Projects in agriculture amount to 4.4 per cent. Energy industries dominate (more than 60 per cent) (UNFCCC 2010).

[27] Annex B countries are those that have obligations to reduce emissions according to the protocol. They are industrial countries and some countries in transition.
[28] Countries may still sell or auction these rights to firms etc. EU/Norway does this for a fraction of the cap.

A main reason why forests are not competitive under CDM relates to the high transactions costs relative to other options. High transaction costs relate again to property rights issues (e.g., Kerr et al. 2004; Lipper and Cavatassi 2004, Cosbey et al. 2005) and the generally high demands set on a country's institutions to make trade in forest carbon possible. Because of this, Africa is generally very little involved in CDM – less than 2 per cent of CERs – while China (51.1 per cent and India (17.7 per cent) top the list (UNFCCC 2010). The rather high level of controversy around some forest projects should also be noted.

The literature on CDM is quite instructive regarding various motivational issues. The key motive for being involved in CDM is to find cheaper solutions to reduce emissions below the cap. Hence, as opposed to PES where the motive is to pay for a better service, CDM is a way to reduce costs for reaching a target. This seems to influence the motivations of those involved. Sutter (2003); Olsen (2007); and Olsen and Fenhann (2008) show how this tends to create 'a race to the bottom' in the sense that going for the cheapest carbon has weakened the ability of CDM to deliver well on the sustainability criteria, like biodiversity protection.

In relation to this, the issue of additionality is strongly emphasized in the literature. There is an incentive among both the buyers and sellers to overestimate the effects of a project – e.g., Schneider (2007) and Sovacool and Brown (2009). Hence, the CDM has been much accused of fraud; see also Ostrom (2009) and Spash (2010).

One may ask why such problems are so conspicuous in the case of CDM, but relatively absent in the case of CTFs. Two aspects appear relevant. First, CTFs may represent a more transparent system. Second, CTFs are oriented at funding national parks and other activities to protect biodiversity. Hence, there are no off-sets, compliance issues involved, no credits issued. Because of this, there are fewer motives towards manipulating data on expected impacts. As we have seen, impacts are not even well documented in the case of most CTFs.

As noted, CTFs represent public/voluntary funding. It is interesting to observe that there seems to be a general difference in motivations among those engaged in voluntary carbon markets as opposed to those engaged in CDM. Neeff et al. (2009) document a clear willingness to pay for more than the carbon in the case of the voluntary market. While buyers of CDM credits look for cheap carbon, those operating in voluntary markets are inclined to think more widely as they are acting to support the common good or building consumer trust. In the latter case, they may be looking for projects that can tell a good story – for 'charismatic carbon' (Stenslie 2010). This illustrates how the institutional structure around the trade influences the focus of traders.

Also, CDM demands a sector of intermediaries. Many have been attracted by the size of the market. Lloyd and Subbaro (2008) emphasize that it has caught the attention of many intermediaries that are mainly 'after the money' and the wider aims of CDM have to some extent been sidelined. In relation to this, one should note that the kind of intermediaries involved in a market solution seems to make a difference with regard to performance. Some intermediaries are 'pure' traders, while others – like some NGOs and CTFs – are involved for broader reasons than just earning income from the trades. In the case of CTFs, this seems clearly to be the case. Concerning NGOs we have found no studies comparing their behavior with 'for-profit' intermediaries. In the case of CDM, there is one indication though. The Gold Standard for carbon credits was developed by NGOs to strengthen the emphasis on sustainable development (Stenslie (2010).

Part III: NEW APPROACHES AND FINANCIAL MECHANISMS FOR SECURING INCOME FOR BIODIVERSITY CONSERVATION

by

David N. Barton, Henrik Lindhjem, Irene Ring and Rui Santos

In this chapter we expand the analysis into a discussion of more novel financial mechanisms – i.e., mechanisms that have not been in much, if any, use so far in the area of biodiversity protection. This will concern solutions like PES procurement auctions, offsets/ habitat banking, and fiscal transfers. Examples of auctions of watershed protection contracts are pilot experiences from a small number of research projects in developed countries; habitat banking/offsets are found mostly in US wetland conservation; ecological fiscal transfer mechanisms financed using value added taxes have been tried in Brazil. Other types of ecological fiscal transfers have been tried out in Portugal and Germany. PES procurement auctions are primarily proposed to reduce the costs of achieving environmental objectives, potentially freeing up resources. Reforming environmentally harmful or otherwise ineffective subsidies which we also discuss in this part, is not a new approach as such, but is still a crucial component of any future mix of instruments to increase potential financing for conservation and create more appropriate incentives.

The extent to which these examples and specific experiences are transferable to other countries depends on the institutional setting in which the instruments were introduced. An analysis of whether they are generally applicable will mainly be based on theoretical reasoning and summary descriptions of the particular policy and development settings in which financial mechanisms have become a part of some countries' policymix. The literature reviewed for Part III is somewhat narrower than in Part II, stemming from consultancy reports and peer reviewed papers, mainly from the field of ecological and environmental economics. We try, however, to use the same set of criteria as used in that part.

Rui Santos is co-author of Section 2 and Irene Ring is lead author of Section 3 of this part of the report.

1. PES PROCUREMENT AUCTIONS

1.1 WHY AUCTIONS?

Auctions have been used by governments to trade a range of different commodities, including e.g. electricity, broadcast spectra, emission permits etc. Auctions come in many forms and the main idea is to harness market forces to induce participants in the auction, the bidders, to compete and through the bidding process, reveal their true valuation of the auctioned good. The underlying challenge for regulation, and the rationale for use of auctions in funding conservation, is that the landowner knows more about the on-site costs and local impacts of various activities than the conservation agency. This so-called asymmetric information gives rise to two problems normally termed hidden information ('adverse selection') and hidden action ('moral hazard') (Latacz-Lohmann and Schilizzi, 2005).

Hidden information may lead to a tendency where landowners who already are engaged in environmentally friendly practices and have low costs of additional such activities, enroll more often in PES programs. This may result in low environmental benefits (low additionality) and overcompensation of compliance or opportunity costs. Hidden action refers to the incentives contracted landowners have not to comply with the contract terms, if compliance monitoring is costly or difficult. Landowners can utilize their private information to extract so-called 'information rents' from the conservation agency, which implies higher costs to achieve a given environmental objective.

With the increased realization that funding for biodiversity conservation is far below what is required, auctions have increasingly been considered as the most promising mechanism to deal with the information asymmetries and increase cost-effectiveness of publicly managed PES programs.[29] Australia and the US are the pioneering countries testing auction designs in conservation and agri-environmental schemes (see e.g. Stoneham *et al.*, 2003).

1.2 TYPES OF AUCTIONS AND EXAMPLES

The two main forms of price setting in ordinary PES schemes to date are (1) bilateral bargaining between the conservation agency ('the buyer') with a single (or group of) landowner(s) ('the seller') where a price is agreed; and (2) posted fixed-price payment schemes. The latter is e.g. used in the Costa Rican PES scheme and in EU agri-environmental programs to support biodiversity (Rousseau and Moons, 2008).

[29] There are two other mechanisms discussed in the literature to deal with this problem. A coarse approach is collecting information about observable landowner attributes that are correlated with their opportunity costs and establish contract prices based on this (Ferraro 2008). Soil types, distance to roads and markets, forest types etc. are examples. The other mechanism is so-called "screening contracts", where different types of contracts are offered designed to fit with different types of landowners, i.e. so that low-cost landowners only will choose the low-payment contracts and high-cost landowners will choose the high-payment contracts ("reveal their type" through contract choice). Screening contracts are very rare in practice.

In a PES procurement auction, on the other hand, landowners are invited to submit bids (their required payment or compensation to enter into a PES contract) for delivery of the types of conservation activities the conservation agency has specified. Since the ecosystem service and biodiversity outputs are difficult to contract directly – cf. the discussions in Part I – the rewards are normally based on a specific set of activities, i.e. change of land management practices, rather than conservation outcomes (e.g. particular ecosystem services or biodiversity benefits). If these activities or practices are relatively homogenous, the conservation agency can rank bid proposals according to cost, and accept bids until the conservation budget is met. Alternatively, the conservation agency may have a target (e.g. number of hectares) to enroll and will accept bids until such a target is met. If the quality of ecosystem services vary among available land areas, not just the opportunity costs, the conservation agency typically ranks bids based on a combination of the (expected, proxied) environmental benefits and the payment levels (costs), e.g. in an index.

Two well-known auction schemes are the conservation reserve program (CRP) in the USA and the BushTender program in Australia. Descriptions of these auctions and how the auction schemes are organized, for example how they calculate the indices to rank bid proposals (see especially the CRP), are given in textbox III.1 and III.2.

The most common forms of bidding rules are the uniform and discriminative price auctions. In a uniform price auction the winning landowners are all paid the same price, typically the highest winning offer price or the lowest rejected offer. In the discriminative price auction each bidder rewarded a contract is offered their own winning offer bid. The main difference between these two auction formats is the impact on bidding behavior. In the uniform auction, the landowner has no incentive to make a bid above his opportunity cost, since what is paid is independent of the bid. However, the conservation agency will have to pay landowners a compensation which is higher than their revealed, true costs (since all landowners will get the same payment). On the other hand, in the discriminatory auction the conservation agency will pay only what the winning landowners bid, but the landowners have an incentive to inflate their bids since what they get depends on their bid. However, this comes at a cost which is the higher chance of not getting a contract. Hence, the auction mechanism will not be able to eliminate all information rents on parts of the landowners. We will discuss the implications of this for cost-effectiveness below.

Text Box III.1. US Conservation Reserve Program

Land retirement has in particular been important for the US agri-environmental policy. Traditionally, land was retired to improve crop prices or protect the soil, but from the early 1990s reducing environmental damage from agricultural production has come increasingly into focus. The Conservation Reserve Program (CRP) is the largest agri-environmental program in the US. It offers 10-15 year contracts for retirement of land from crop production. To be enrolled in the program land has to have a history of crop production, be highly erodible, and be located in a national or state Conservation Priority Area, or be devoted to wetland restoration, streamside buffers, or conservation buffers. In exchange for land retirement the land owners can receive cost-sharing for establishment of new cover (like grass or trees) on the land, and annual payments to compensate foregone profit and maintenance costs. Land owners who want to participate have to offer bids specifying the land they are willing to give up for retirement, what kind of cover they would establish, and what kind of compensation they will accept. The incoming bids are ranked using the Environmental Benefit Index

(cont.)

(EBI), which includes costs, and the highest scoring contracts are accepted. Prior to the early 1990s all bids under a pre-specified limit was accepted, but this practice has been abandoned to encourage farmers to bid against each other to reduce costs. The EBI factors used to rank bids are related to wildlife, water quality, erosion, enduring benefits, air quality and cost. Land owners may improve their EBI score and thus enhance their chances of being accepted into the program for example by asking for lower annual payments, forego cost-sharing, or establish cover that is more effective as wildlife habitat.

Table III.1 Factors generating points for the conservation reserve program's environmental benefit index

EBI factors	Definition	Features that increase points	Maximum points
Wildlife	Evaluates the expected wildlife benefits of the offer	- Diversity of grass/legumes - Use of native grasses - Tree planting - Wetlands restoration - Beneficial for threatened/endangered species - Complements wetland habitat	100
Water quality	Evaluates the potential surface and ground water impacts	- Located in ground-or-surface-water protection area - Potential for percolation of chemicals and the local population using groundwater - Potential for runoff to reach surface water and the population	100
Erosion	Evaluates soil erodibility	- Larger field-average erodibility index	100
Enduring benefits	Evaluates the likelihood for practices to remain	- Tree cover - Wetland restoration	50
Air quality	Evaluates gains from reduced dust	- Potential for dust to affect people - Soil vulnerability to wind erosion - Carbon sequestration	45
Cost	Evaluate cost of parcel	- Lower CRP payments - No government cost-share - Payment is below program's maximum acceptable for area and soil type	Varies, but commonly 150

Source: Claassen et al., (2008)

The U.S. Department of Agriculture offered a new general signup for the CRP in August 2010, the first since 2006.The details of the EBI calculations for the new program are given in USDA-FSA (2010).

Source: Adopted from Zandersen et al., (2009).

Text Box III.2. BushTender I & II, Australia

The BushTender was initiated by the Victoria government in Australia in 2001. The aim of the tender was to test whether auctions could efficiently purchase public environmental goods from private landholders. The good in question was biodiversity as captured through improved 'bush' management. 'Bush' in Australia refers to the original deep rooted ligneous vegetation prior to clearing and farming, which in agricultural areas survives today usually in isolated patches. Key issues in the initiative was to test how to ensure a sufficient number of landholders participating in the tender and whether an auction could be more cost-effective, budget wise, than a traditional fixed price payment scheme.

Under the BushTender, micro-regions were designated and a budget of A$ 400,000 was allocated in the first round and A$ 800,000 in the second round. Expressions of interests were called for and government officers subsequently visited the farms and the proposed land areas up for tender. Ecological data was collected from the sites to construct a spatially specific biodiversity benefits index, defining a benefit to cost ratio for the government. Contracts were negotiated on a one-to-one basis whereby a land management plan would be set up as a proxy for payment of the ecosystem services. Contract durations were set at 3 years in round 1 and 6 years in round 2. A sealed-bid discriminatory price auction was used to 'reveal' the price of the farmers for providing their pre-negotiated services. Bids were ranked according to the biodiversity benefits index until the budget constraint was hit.

Lessons learnt from the auction were generally positive. The government found that auctions work and contracts are allocated, whereby the marginal cost curve information is revealed and they show improved cost-effectiveness over fixed pricing schemes. The government found that revealing all information on e.g. the biodiversity benefits index to the farmers is best despite the risk of collusion, which they also found was a non-issue. In addition, the government found that auctions are popular with landholders as biodiversity is translated from a complex idea to practical actions. A total of 300 contracts were allotted (Latacz-Lohman and Schilizzi, 2005).

Source: Adapted from Zandersen et al., (2009).

Auctions contain many other design elements than those explained above that may potentially influence bidding behavior and the assessment of auctions as a suitable instrument along our four criteria: process legitimacy, effectiveness, efficiency and equity. We will not go into these design elements in detail but some that may potentially be important for conservation auctions are: whether the conservation agency sets a reserve price[30]; the bid evaluation system; potential for incorporating site synergies; how much information should be given to prospective bidders; whether bids are allowed to be revised and whether auctions are repeated over time (which is typically the case for many conservation auctions in practice). The theory and experience of conservation auctions are relatively immature (Ferraro, 2008). Most of the focus to date has been on estimating potential cost savings of auctions compared to fixed price schemes. Since PES procurement auctions are an alternative to such fixed price PES schemes (as discussed in section 2 in Part II), we emphasize below differences involved in using auctions.

[30] An upper limit on the amount of the agency is willing to pay for a unit of the conservation good.

1.3 PROCESS LEGITIMACY

In the rapid literature review we have done for this assessment, we have found no studies investigating process legitimacy of auctions over fixed-price or bilaterally negotiated PES contract schemes. It is impossible to make general judgments of process legitimacy of auctions vs. standard PES schemes, as this is likely to depend very much on the local context and the specific type of auction design. However, we mention a few important and general auction design issues here that may be of importance for how actors judge the degree of process legitimacy.

First of all, most if not all conservation auctions to date have been organized by a public entity, leaving most of the decisions and judgments to this entity. For private actors to run an auction it would require a cap on environmentally harmful activities, which could give private actors an incentive to reach the cap at lowest cost through an auction mechanism (see discussion in Part I and the section on habitat banking below). The alternative would be for private actors to be voluntarily involved in auctions. To our knowledge privately organized auctions for conservation contracts have not yet been used, likely since it is just government entities that are willing to be the buyer of conservation contracts. Hence, the rights structure underlying most current auctions reflects that the landowners have the right to the current level of deterioration and will get compensated to provide environmental benefits beyond this level (see Part I).

Decisions of eligibility to participate, the information given to bidders, bid evaluation system and how environmental benefits are weighted and scored compared to costs, bidding rules and how many bidders are finally given contracts are all design elements that may be more or less transparent and judged as more or less fair and accountable by different stakeholder depending on the situation. For example, the decision to differentiate bid prices using a discriminatory auction may be rendered suspect and prone to corruption compared to a uniform price auction, if the criteria under which price differentiation have been decided are not transparent. The choice of bidding rules is also important for equity considerations (see below). Another example of a design element of importance for process legitimacy is given by Cummings et al., (2004): If landowners are allowed to revise their bid during the auction process to reduce the chance of poor choices, the likelihood that landowners will be angry about the auction process may be reduced.

Many of the design elements we mention also have more or less clear implications for cost-effectiveness and efficiency of the auction, and it is likely here as in the choice of other policy instruments, that there will be trade-offs between different criteria. For example, although it may be desirable for process legitimacy to give as much information as possible about an auction process, this information may be used by landowners to extract information rents and thereby reduce the cost-effectiveness of the auction mechanism, especially in auction processes that are repeated (Latacz-Lohmann and Schilizzi, 2005). Further, if sophisticated bid evaluation methods are used to combine environmental benefits and costs, this process may be perceived as a ‘black box’, is harder to explain to bidders and generally less transparent than other pricing schemes. Such evaluation processes are also by necessity subjective and may be more prone to manipulation and rent extraction by officials.

Another issue that Latacz-Lohmann and Schilizzi mention that may be potentially important for process legitimacy is whether the contract is based on input activities or environmental outcomes. If the contract is based on outcomes, land owners take all the risk to ensure such outcomes are achieved. If they are hard to observe or measure, there may be high risk of disputes (e.g. litigation) linked to unclear landowner responsibility.

1.4 EFFECTIVENESS AND EFFICIENCY

The main question related to *effectiveness* in this case is whether auctions can deliver reduced biodiversity and ecosystem service loss over the more standard PES-schemes. As with auctions, standard PES schemes are subject to hidden action problems, i.e. that the landowners have an incentive to cheat on their contract obligations resulting in lower-than expected environmental benefits. The conservation auction literature is not well-developed to deal with this problem. The most important policy conclusions in this literature are derived from theory and simulations. According to Latacz-Lohmann and Schilizzi (2005) the incentives to cheat are expected to be highest when landowners' compliance (opportunity) costs are high in relation to the payment levels (and the detection probability and fine in case of non-compliance detection are both low). This means that overcompensation may reduce the risk of non-compliance (and therefore the need to monitor) and that monitoring efforts should be concentrated on the high-cost landowners. These are the landowners with high pre-contractual land-use intensities.

The impact on effectiveness of auctions compared to fixed-price PES schemes is hard to judge on this point, as auctions typically contract more high-cost, high environmental benefit landowners who also have a higher risk of non-compliance. If monitoring for these landowners is not increased, some of the additional environmental benefits may not be realized compared to a fixed-price PES scheme. However, as argued by several authors, the chance of achieving additionality should be higher overall with the use of an auction (Ferraro, 2008). That is because paying low-cost landowners less through an auction frees up resources to pay high-cost landowners, who are much more likely to provide a (much) lower level of ecosystem services in the absence of a PES contract. This is a potentially important point, given the low additionality observed for example for parts of the Costa Rican PES scheme (see Part II of the report).

Auctions may also be more targeted to take account of the heterogeneity of ecosystem services over the landscape, not just variations in opportunity costs. This may be done by separating auctions with groups of landowners that are relatively homogenous in the services they supply or by scoring environmental benefits using some form of index, as in the CRP scheme. This brings us back to the problem of how to value or weight environmental benefits. However, even some form of consideration or weighting of environmental benefits may yield significant efficiency and environmental improvements over auctions that only consider the heterogeneity in costs (Claassen *et al.*, 2008; Connor *et al.*, 2008). Auctions may also be designed to encourage bids (and higher payments) for land areas that are contiguous, giving higher environmental benefits than similar-sized plots away from each other (Reeson, forthcoming). However, auction design must carefully consider the risk of so-called collusion and strategic bidding among landowners which will reduce cost-effectiveness. Concentrating contracts in one geographic area may also reduce monitoring costs, and in some cases landowners may influence each other positively, reducing the likelihood of breaching contract obligations.

Whether contracts are based on inputs (e.g. prescribed management activities) or outcomes (i.e. some measureable part of the final ecosystem service or biodiversity change) – or a combination of the two – is important for effectiveness. If the contracts are based on input activities only with no reference to achieved outcomes, landowners will have no incentive to make sure the outcomes are achieved or for entrepreneurship, e.g. providing biodiversity habitat more cheaply.

We will briefly discuss other aspects of effectiveness, e.g. the risk of corruption, in the final section below.

The main issue analyzed in the literature is the potential cost savings of auctions compared to standard fixed-price PES schemes, i.e. *cost effectiveness*. The general view in the literature based on actual experience and model simulations is that auctions may reduce costs of reaching environmental objectives substantially compared to fixed-price PES arrangements (Ferraro, 2008; Rolfe and Windle, 2008; Windle and Rolfe, 2008). It varies whether these studies incorporate some measure of administrative or transaction costs. Auctions seem to work best when there are many bidders, contracts are fairly homogenous in the ecosystem services or input activities, and landowners are heterogeneous in their opportunity or compliance costs (Latacz-Lohmann and Schilizzi, 2005).

Latacz-Lohmann and Schilizzi (2005) recommend using a discriminatory bidding rule rather than a uniform one for conservation auctions if there are no clear reasons to think that bidders will grossly inflate their bids over opportunity costs. If the landowners are risk-averse and prefer a certain income from a PES contract over more uncertain alternatives, the discriminatory format may clearly be preferable to the uniform format. This is because land owners would tend to bid less as they also value a more secure income. Note also that a uniform pricing is likely to give lower overall payments than a fixed-price scheme for the same environmental target.

A final issue we will discuss regarding cost-effectiveness is the role of information and learning. Although many studies show large efficiency gains of auctions over fixed-price schemes, these studies have typically assessed one-shot auctions. More recent studies show that if landowners learn from previous auctions or get information for example about the specific preferences of the conservation agency or their maximum reserve price, these efficiency gains may be greatly reduced over time (and space) as landowners adjust their bids upwards (Schilizzi and Latacz-Lohmann, 2007). As mentioned, in this regard information sharing to achieve higher legitimacy may have to be traded off with increased cost-effectiveness.

Regarding *transaction costs* of auctions, few studies investigate this issue explicitly. Auctions can be complex and difficult to implement and therefore imply transaction and administrative costs that are higher than fixed-price schemes (e.g. Connor et al., 2008). However, Ferraro (2008) argues that auctions may not be more complex than individually negotiating with landowners, which some countries do. Using differentiated payments, e.g. as a result of a discriminatory price auction, may in turn be more administratively costly than a uniform price auction. It would also involve potentially higher transaction costs to attempt environmental outcome-based contracts (Latacz-Lohmann and Schilizzi, 2005) or environmentally heterogeneous auctions where benefits need to be weighted. As mentioned above, environmental outcome based auctions may lead to litigation and conflict as landowner responsibilities, for example in case of a forest fire, pest or similar, may be less clear.

1.5 EQUITY

As we have seen, auctions may take many forms and the outcomes will depend on the specific design of the auction. Compared to a fixed-price PES scheme, payments will generally be made to fewer landowners, a larger percentage of who are high-cost (and high environmental benefit). This may have equity and poverty impacts, if low-cost landowners are more likely to be poor. Further, in some countries it may be considered more equitable to make differentiated payments that reflect opportunity costs, rather than uniform payments to all landowners. However, in developing country contexts, uniform payments are likely to be seen as more fair (Ferraro 2008). It may not be regarded as fair, that those who have demonstrated environmentally friendly behavior in the past and therefore may have lower opportunity costs are ‘punished’ for that with lower payments. This has been a key issue in the debate about REDD. As many PES schemes in reality have both poverty alleviation and conservation objectives, there may be additional trade-offs between equity and efficiency for PES auctions in such contexts.

1.6 CONCLUSIONS AND TRANSFERABILITY TO DEVELOPING COUNTRY CONTEXTS

Conservation auctions are still in their infancy, even in industrial countries, though the interest and experience are growing fast. Auctions can potentially save substantial costs to reach environmental objectives compared to fixed-price PES schemes or individually negotiated PES-contracts when there are many bidding landowners (encouraging competition), contracts are fairly homogenous in the ecosystem services or input activities, and landowners are heterogeneous in their opportunity or compliance costs. However, the complete transaction costs have rarely been calculated comprehensively in practical auction applications.

While auctions are institutionally, technically and informationally relatively demanding to organize, they may still be feasible in low and middle-income countries, as many of these countries already have relevant experience e.g. from relatively simpler timber and forest product auctions (Ferraro 2008). However, there are practical differences between very low income countries, where the conservation agency will have to contract with landowners that are very poor, semi- or illiterate, without formal land title and dispersed over remote, rural areas. Auctions will be practically and institutionally easier for a mid-income country such has Costa Rica.

Ferraro (2008) points out that the use of differentiated payments may provide opportunities for private gain and corrupt activities such as bid rigging, potentially reducing the effectiveness of auctions as an instrument to achieve higher biodiversity gains at lower costs. This will be a more serious problem in low income countries.

Finally, Latacz-Lohmann and Schilizzi (2005) warn in their review of the suitability of using auctions in Scottish agri-environment PES schemes that auctions due to their complexity (and often unpredictable outcomes) may have a higher risk of failure than a fixed-price scheme. Hence, more testing is likely to be required before conservation auctions are rolled out, especially in developing country contexts.

2. TRADABLE DEVELOPMENT RIGHT AND HABITAT BANKING WITH BIODIVERSITY OFFSETS

The discussion of habitat banking with tradable development rights and biodiversity offsets draws mainly on a review conducted by the POLICYMIX project[31] (Santos *et al.*, 2011a)[32]. In addition, we have reviewed findings from regional reviews for Latin America and the EU (Bovarnick *et al.*, 2010; EFTEC *et al.*, 2010), business manuals (BBOP, 2009), and some peer reviewed literature (Karsenty, 2007; Benayas *et al.*, 2009; Hartig and Drechsler, 2009, 2010). In depth case studies of existing biodiversity offset schemes are available as appendices to EFTEC et al., (2010). Bovarnick et al., (2010) conduct a feasibility assessment of habitat banking for selected countries in Latin America which are available as appendices to their main report.

In this chapter we discuss the difference between tradable development rights (TDR) and habitat banking with biodiversity offsets, consisting crucially of whether a development cap or conservation objective is the object of the trades/offsets. Biodiversity offsetting does not require trading, and can be based on a pure command and control approach (obligation to compensate under environmental liability either on- or off-site). Habitat banking is a particular case of biodiversity offsets, introduction a trading element for offset actions (credits) that are delinked both in space and time from the specific development requiring compensation (debits). Both TDR and biodiversity offsets are created by legal imposition of either development cap/zoning requirements (TDR) or environmental liability/protection legislation requiring off-setting of residual impacts of development, respectively. In practice, legislation mandating biodiversity offsets has been harder to implement with biodiversity offset and banking characterized more as ‘pilot’ or ‘voluntary’ projects (see e.g. French and Australian case studies, EFTEC et al. 2010).

Trading is the basic feature associated with both TDR and habitat banking (market-based instruments): in one case trading ‘development units/rights’ in the other trading ‘conservation/ habitat units/areas’. The essential difference is the definition of what is traded, and the information required to achieve equivalence between the sites involved in the trade. On the one

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[31] http://policymix.nina.no.

[32] Santos and colleagues review biodiversity offsets, tradable development permits and habitat banking for biodiversity conservation. Although the specific purpose of their review is to assess the complementarities of offsets, permits and banking relative to other instruments in the conservation POLICYMIX, we use their review as a starting point for our own discussion. They follow a similar evaluation format to the present report, first reviewing process related issues of governance levels and actors involved, setting the baseline for offsets, monitoring and evaluation; experiences with policy outcomes criteria, namely effectiveness, cost-effectiveness, social impacts and institutional and legal requirements.

hand, TDRs do not require an environmental mitigation hierarchy to be implemented, and trade in an observable proxy of the quality of development (e.g. infrastructure density). On the other hand, habitat banking with biodiversity offsets bases the trade on habitat quality as a proxy indicator for biodiversity; conditioned by legal requirements to carry out as much mitigation of development as possible at the development site. While trading is a superficially similar characteristic, there are large differences in principle between how policy effectiveness is defined and the potential transaction costs of achieving development versus conservation policy targets.

We focus our review on some assumptions made by studies regarding the trade-offs between effectiveness, costs and equity.

2.1 DEFINING CHARACTERISTICS OF TRADABLE DEVELOPMENT RIGHT AND HABITAT BANKING WITH BIODIVERSITY OFFSETS

Tradable development rights (TDR) rely on a cap of the total amount of development allowed in an area identified for its conservation value (EFTEC et al., 2010). According to EFTEC (2010: 43) "TDR programmes separate out the right to develop land from other rights such as use and lease. As the right to develop land is sold, that parcel of land becomes protected from development, often as a conservation easement. The parcel of land that the rights are transferred to is then allowed to develop, in some cases to a higher degree than normally would be allowed by standing planning permission." An example is Brazil's Forest Code requirement of 20% set aside of land in legal reserves (35% in the Amazonian cerrado habitat, 80% in the legally defined forested Amazon). Slightly confusing is that 'offsets' is also used in the literature when explaining the compensation requirement in Brazil's Forest Code (EFTEC 2010, appendix). The difference in TDRs and habitat banking with biodiversity offsets is explained below.

TDR is a way of encouraging the reduction of development in areas that should be protected ('sending zone') and increasing development in predicted growth areas ('receiving zones') (Pruetz, 2003). Demand for TDR's is created by regulation of a cap on development or a minimum reserve requirement (Figure III.1). It is also a pre-condition for a market that there is a difference in the opportunity costs between the location seeking to purchase the TDR (A) and the off-set site (B). We return to other preconditions in our discussion below. Experiences with TDRs can be found in countries such as the USA and Brazil where property rights have been defined as a bundle instead of a single right (EFTEC et al., 2010). Where public money is used to purchase rights, rather than the purchase occurring between private developers and conservation, this is referred to in the USA as Purchase of Development Rights (PDR).

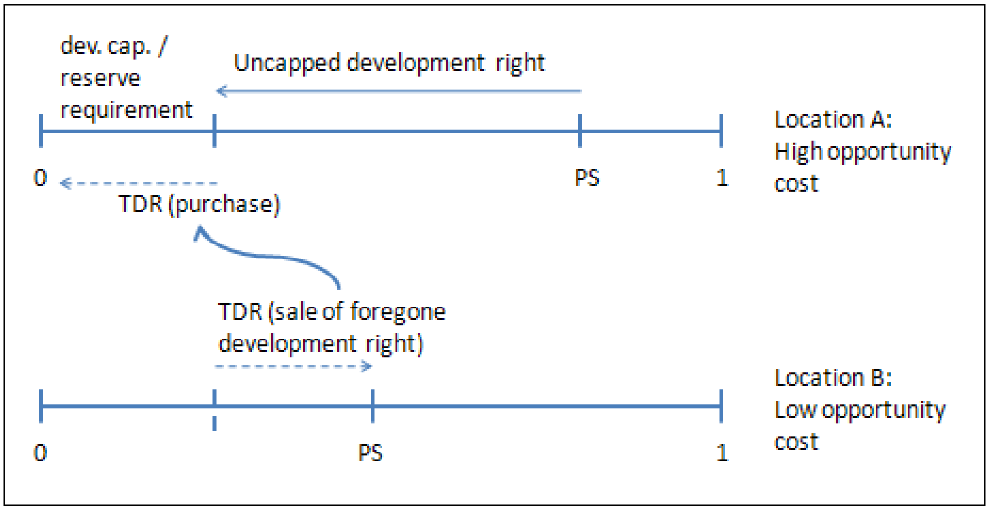

Figure III.1. Tradable development right approach. PS=present status.
Owner of location B ('sending zone') relinquishes her right to develop land, selling it to offset landowners A's ('receiving zone) liability in exceeding a development cap.

The terms-of-trade reflect both the polluter pays principle (PPP) on the part of the developers liability (in the form of PPP2 as defined in Part I), and the provider gets principle (PGP2), on the part of the rehabilitating landowner – tradable development rights refer to the rights a landowner (B) has above the development cap to place her land in a conservation easement, which is then sold to offset the liability of a developer (A) who exceeds a development cap.

Biodiversity offsets are defined by the Business and Biodiversity Offsets Programme (BBOP, 2009:6) as "measurable conservation outcomes resulting from actions designed to compensate for significant residual adverse biodiversity impacts arising from project development and persisting after appropriate prevention and mitigation measures have been implemented. The goal of biodiversity offsets is to achieve no net loss, or preferably a net gain, of biodiversity on the ground with respect to species composition, habitat structure and ecosystem services, including livelihood aspects".

Biodiversity off-sets are defined by environmental liability of development (Figure III.2). A developer's initial project plans may entail environmental liability being subject to a mitigation plan following an Environmental Impact Assessment (EFTEC 2010, BBOP 2009). The mitigation hierarchy regards biodiversity off-sets as a measure 'of last resort' after having taken prior mitigation steps to 1. avoid; 2. minimize; and 3. mitigate impacts on-site. Off-sets are meant to compensate for the residual on-site impact after these measures have been implemented.

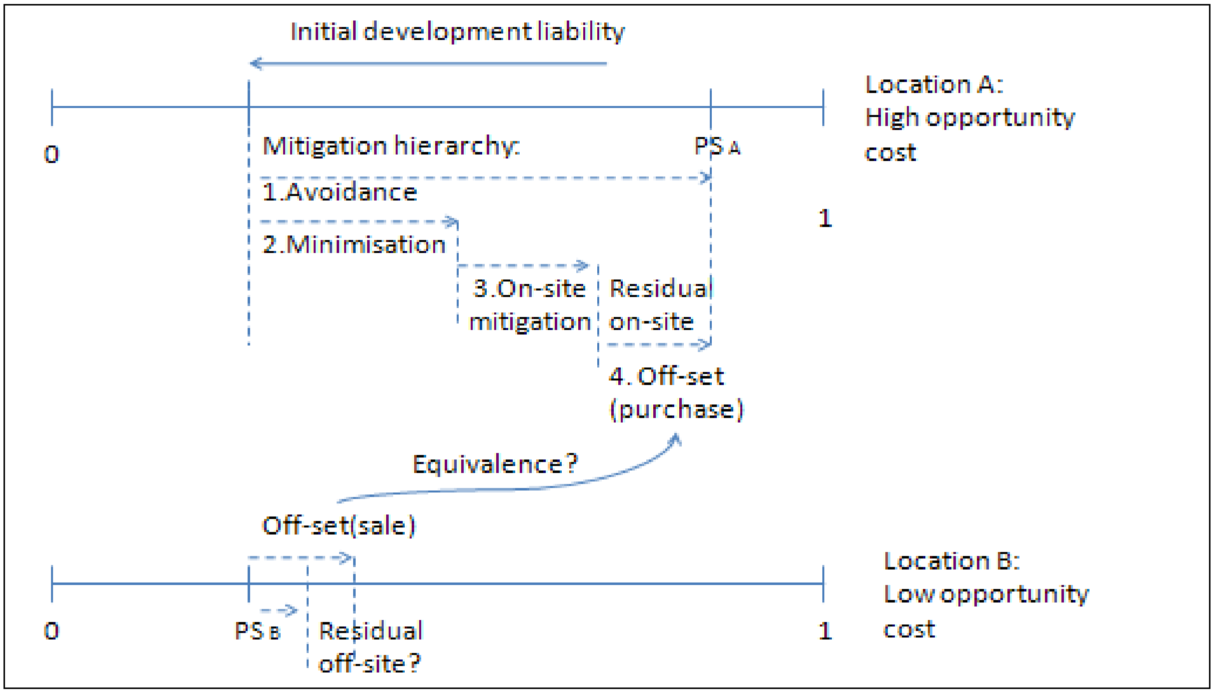

Figure III.2 Biodiversity off-set concept and issues.

'Residual on-site' refers to unmitigated development which requires a purchase of an off-set to achieve 'no net loss'. 'Residual off-site' refers to the possibility that the off-set credit sold from rehabilitation off-site may not actually achieve the contracted off-set because of ineffective rehabilitation. Equivalence also refers to similarity of habitats' composition, structure, and function their ecosystem services

In figure III.2 off-sets are purchased by site A developer either directly, or through a habitat banking arrangement, from a site B where a habitat restoration activity is undertaken, starting from the present status (PS), that aims at creating equivalent habitat to that lost. The land under habitat restoration at site B is placed in a conservation easement to ensure its future protection.

The equivalence of rehabilitation is key to the effectiveness of off-sets in attaining a 'no net loss' target. If more off-sets or off-sets of a better habitat quality are purchased this is termed 'trading up' (EFTEC et al., 2010). On the other hand, lacking enforcement at the rehabilitated site B can lead to less than equivalent rehabilitation, in which case the residual biodiversity impact of the development activity is effectively 'exported' off-site (Figure III.2). A clear definition of residual biodiversity loss requires that developers' obligations and liability under a mitigation hierarchy are clearly defined, understood and enforced. The effectiveness of biodiversity off-sets relies on equivalency of credits between habitat types and on enforcement of biodiversity protection.

Habitat banking is also referred to as *biodiversity trading*, *biodiversity banking* and *conservation banking* in the literature. Habitat banking is defined by EFTEC et al., (2010:62) as "a market where credits from actions with beneficial biodiversity outcomes can be purchased to offset the debit from environmental damage. Credits can be produced in advance of and without ex-ante links to, the debits they compensate for, and stored over time". The 'banking' aspect relates specifically to biodiversity off-sets being stored and unrelated *ex ante* to the development creating the debits. By decoupling the development activity from the off-setting activity, habitat banking holds out the prospect of matching low cost land for rehabilitation to high value land for development. However, the decoupling in time of the compensating action by using a habitat bank emphasizes further questions regarding habitat equivalence between credit and debit activities at different sites.

2.2 Policy and design issues

Although the definition of habitat banking can be applied to TDRs, we find most examples of habitat banking in the literature referring to biodiversity offsets (EFTEC 2010 appendix): most states in Australia have introduced some form of banking with biodiversity offsets, such as 'native vegetation offsets' in the state of Victoria sold through a government regulated trading system called 'Bushbroker'; New South Wales has regulations allowing mitigation banking through its 'Biobanking' scheme; South Africa has a wetlands mitigation banking system; USA has wetlands mitigation banks and conservation banks, the latter striving to mitigate adverse development effects on species listed as threatened or endangered under the Endangered Species Act; Canada's Fisheries Act requires compensation for lost fish habitat prior to the development of a project, often used in situations where a developer needs to compensate for several small harmful alternations, disruptions or destructions, but there are few on site compensation options.

It has been argued that development rights and biodiversity off-sets in habitat banking schemes are expected to perform better than landscape zoning without habitat banking in terms of cost-effectiveness and private actor engagement (Santos et al., 2011a). Extensive review of the conditions under which habitat banking is both feasible and effective was conducted by EFTEC et al., (2010). They list the potential major benefits from habitat banking (p.5 Executive Sum.):

- More effective, and in some cases ex-ante (and therefore more reliable), delivery of existing biodiversity policy objectives and of compensation requirements;
- Greater impacts and increased long-term viability of large-scale measures (also potentially from pooled offsets);
- Reduced habitat fragmentation from strategic and selective placement of compensation measures (e.g. to link up, increase the size of, or buffer Natura 2000 sites);
- The option to trade up measures to address higher conservation priorities, and
- The opportunity to efficiently address cumulative impacts from small-scale or low impact developments for which there is no legal requirement for compensation.

The listed benefits of habitat banking are relative to other market-based instruments – rather than a status quo without habitat banking (and based largely on regulation). The analytical starting point of the EFTEC analysis would seem to be that a decision on some kind of market-based instrument has been taken. On the contrary, we argue below that an analysis relative to a status quo, or alternatives based largely on regulation is missing. We also argue that conclusions regarding feasibility of habitat banking have been made assuming that an effective scheme can be designed. There is reason to question such an assumption. Recent reviews of mitigation banking schemes have shown high rates of non-compliance with agreed conditions (Gibbons and

Lindenmayer, 2007 in Santos et al., 2010a). A review of rehabilitation projects from around the world also concluded that rehabilitation projects fell well short of reference ex ante levels of biodiversity (Benayas et al., 2009). In the next sections we look at some explanations.

2.3 LEGITIMACY OF THE POLICY PROCESS IN ESTABLISHING HABITAT BANKING

Legitimacy of the process concerns how decisions are made, who participates and under what conditions. Part II discussed the legitimacy of market versus public governance in the context of PES as a conservation instrument. In the rapid literature review we have done for this assessment, we have found no studies investigating process legitimacy of habitat banking over other market mechanisms. It is difficult to make general judgments of process legitimacy of habitat banking or TDR versus other instruments, as this is likely to depend very much on the local context. Here we discuss how habitat banking, while a market mechanism, relies on public governance to create market-demand by defining rights, monitoring and enforcing compliance of the terms of mitigating and offsetting habitat loss.

Setting development caps, biodiversity conservation targets and environmental liability levels are political decisions and keys to establishing markets for habitat banking. Demand for off-sets comes from the business sector interested in development of natural areas, but is caused by the cap set by the policy process. Politically complex, translating international targets to national/regional/local level conservation objectives is steeped in politics, but also essential to local habitat banking markets. The main constraint on habitat banking in countries such as France is the absence of standards at the national level (CDC Biodiversité 2010 in Santos et al., 2011a). In the EU, the feasibility of habitat banking within member states rests on policy reforms in these states to implement the EU's 2020 Biodiversity Strategy objective of 'no net loss' of biodiversity. In another example from Brazil, the feasibility of a tradable development rights scheme rests on the definition of the percentage of land in legal reserves required of landowners in the presently disputed Codigo Floresta. Here, individual states also have autonomy in defining how the cap is to be implemented (EFTEC 2010, appendix pp. 111-129).

While the focus on the literature is on equivalence and the effectiveness of the biodiversity off-sets *off-site* (location B in Figure III.2), there is an incentive *on-site* to increase what is unremediable 'residual biodiversity loss'. With lacking enforcement developers have an incentive to substitute more expensive on-site minimization and mitigation measures for cheaper off-site biodiversity off-sets. Where this happens it could damage the legitimacy of habitat banking. Habitat banking therefore relies on political support for strengthening *enforcement of the 'mitigation hierarchy'* following from Environmental Liability and SEA.

The potential size of the market for habitat banking is a political question of the definition of rights and responsibilities of public and private actors, defined by environmental liability and requirements of compensatory measures. EFTEC et al. (see previous comment), (2010) conduct an extensive review of legislative basis for habitat banking in the EU Habitats Directive and the Environmental Liability Directive. They conclude that with current EU legislation the potential for habitat banking in Member States is small, but potentially large for 'less protected' and 'widespread' habitat types (Figure III.3).

	I. Critical	II. Strictly protected (A)	III. Less protected (B)	IV. Widespread (C)
Legal status	EU Laws & Directives		National policy priorities	Limited
Compensation driver	- n/a	Habitats & other Directives - Guidance	Weak - planning laws	None
			New mechanism **required to ensure no net loss**	
Potential market	None for debits	Small	Currently small, but potentially large	
Equivalence approach?	-	Detailed, case by case		Simple checklist, possible fee
Equivalence like for like?	Trading up to credits	Strict	Strong	Weaker (trade up)

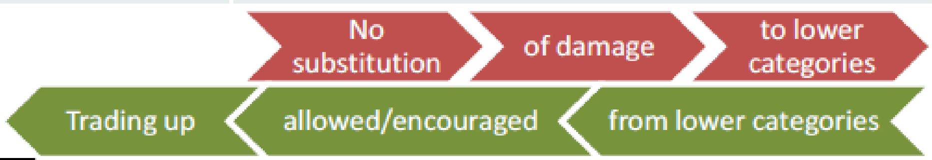

Figure III.3 Different aspects of habitat banking according to conservation status of the biodiversity involved (Source: EFTEC et al., 2010)

While Figure III.3 illustrates that the potential for a large habitat banking market is only for less protected and widespread habitats, these also provide the least valuable biodiversity off-sets in terms of habitat value. The EU Habitats Directive regulates the use of protected areas in the Natura 2000 network. None of the reviews of habitat banking we have looked at conclude that protected areas should be opened for generating off-sets. Nevertheless the Habitats Directive is currently seen as an obstacle to expanding off-sets to strictly protected areas. EFTEC et al's. (2010:199) legal evaluation illustrates this motivation "[..] the European Commission makes clear that compensatory measures should have a strict connection with the affected habitat type and its functions. It is this requirement that in our view poses the largest obstacle to habitat banking. The question arises how to circumvent this obstacle". It is further noted that "In order to use habitat banking as an alternative to the compensation measures that have to be taken on the basis of Article 6(4), we consider adjusting the Habitats Directive as necessary, especially *when the aim is to use habitat banking on a large scale* " (op.cit. p. 127).

As the last quote would seem to indicate, there is a danger that implementation of market-based instruments in conservation policy is taken as the policy aim in itself, rather than a cost-effective means of attaining conservation aims. Where this is the case there is an incentive for off-set brokers and developers to push for less strict compensation requirements.

We now turn to arguments relating to the evaluation criteria of effectiveness, costs and social impacts of habitat banking.

2.4 LEGITIMACY OF OUTCOMES

The main aim of this section is to illustrate how several key design aspects of habitat banking involve hard trade-offs between evaluation criteria.

2.4.1 Effectiveness and efficiency

As defined in part I *effectiveness* concerns the capacity of habitat banking to deliver reduced biodiversity loss and the capacity to ensure additionality and permanence and avoid leakage. Implicit is also how the system influences motivation – including the risk of corruption.

Costs include the opportunity costs of foregone development income due to conserving the sites, and transaction costs of ensuring effectiveness of caps and mitigation measures on the development site, and off-setting compliance at the rehabilitation site. Transaction costs are also involved in establishing the regulatory conditions for trading in habitat banking.

Efficiency is defined as achieving the highest effectiveness at the least cost (independently of a target). A related concept is cost-effectiveness, which means achieving a set target at least cost. Cost-effectiveness is the most relevant for our discussion when conservation targets (such as 'no net loss') are explicit. We also discuss effectiveness and efficiency/costs together as they are in many instances correlated with each other. In evaluating cost-effectiveness of habitat banking we would argue that much of the review literature makes claims regarding based on theoretical expectations, and not in relation to any particular alternative market-based instrument, let alone a reference situation not involving market-based instruments. For an example see Table III.2 (EFTEC et al., 2010)

Table III.2 A comparison of habitat banking and other market based instruments. EFTEC et al., (2010:4) writes "This favourable comparison [of habitat banking to other MBIs] is contingent on it being possible to design an efficient system, which balances regulatory controls of risks with freedom for the market to operate." In other words, the conclusions of greater cost-effectiveness of habitat banking are based on an assumption of efficient design! Our approach in this chapter is limited to a discussion of design issues, unfortunately with little empirical evidence to draw on. From the list of issues discussed in Santos et al., (2011a) we have selected some particularly difficult trade-offs that must be considered in finding the balance between costs and effectiveness of habitat banking[33].

[33] What is cost-effectiveness of habitat banking to society? Cost-effectiveness is a ratio of effectiveness in achieving a "no net loss" target relative to social costs of establishing the system. Effectiveness can be represented by the net gain in habitat area (A) and quality (Q) summing across at the rehabilitation site (b) and the development site (a). Costs include the additional transaction costs (TC) of complying with the conditions of the biodiversity off-sets credit at both locations a and b. There are also information costs (i) of habitat bankers identifying owners with

Table 2.1: Comparison of habitat banking to other relevant market based instruments

	Instrument		Economic Rationale			Environmental Effectiveness	
Type	**Theory**	**Practical Issues**	**Burden**	**Gain**	**Efficiency**	**Effect**	**Long-term**
Habitat banking Features	• Polluter pays • Can deliver fixed policy objective (e.g. no net loss), but cost (price) can fluctuate.	• Careful design of system essential, especially rules on equivalence, monitoring and evaluation • Over designed market may not function	• Private finance • Successfully implements polluter pays • Risk of non-additional actions	• Avoid biodiversity loss • Possibilities for Trading up[11] or other strategic objectives	• Economies of scale at several stages of compensation • Potential financial and ecological benefits • Reduced transactions costs	• No net loss • Potential for net gain • Incentive to conserve biodiversity • Difficult to assess for long-term credits	• Direct resources to conservation priorities (e.g. valuable habitat or climate change adaptation)
Comparison of habitat banking (HB) to other MBIs	Favourable HB has fixed objectives (NNL), but price fluctuates - appropriate to heterogeneous resource like biodiversity and thus likely more efficient for biodiversity than a tax-based solution.	Acceptable • Potential problems shared by HB and other instruments targeted at biodiversity	Favourable • No additional cost to public sector (other than regulatory costs, which can be recovered from HB providers). • Minimal deadweight loss • Competition minimises prices	Favourable • HB gives individuals incentive to go beyond minimum compensation requirements • Design for biodiversity policy needs possible.	Acceptable • HB creates market incentives at several stages of biodiversity conservation process • Detailed design and oversight may raise transactions costs.	Favourable • Potentially creates efficient system for delivering compensation requirements • Environmental outcome fixed at baseline (no net loss) (in theory)	Favourable • Mechanisms to ensure permanence can be built into system • Unclear incentives for long term monitoring

Source: EFTEC et al., (2010)

In the next section we discuss how achieving habitat equivalence is constrained by available habitat for offsets, differences in opportunity costs between sites, private motivations to substitute away from the costly like-for-like habitat offsets, and the transaction costs of the regulator in trying to guarantee that offsetting actually happens and 'no net loss' conservation targets achieved. We divide our discussion into four main issues (i) 'Like for like' – quality and substitution (ii) opportunity costs, (iii) Transaction costs and (iii) Motivational aspects

(i) 'Like for like' – quality and substitution

Achieving habitat equivalence between developed areas and offset areas is synonymous with the conservation effectiveness of habitat banking and TDRs.

Piecemeal versus functional conservation. If large contiguous areas of unrehabilitated land are not available next to existing intact habitats, and/or landowners there are not interested in giving up their development rights, a piecemeal rehabilitation of habitat 'islands' could result, and the potential for habitat banking to agglomerate rehabilitation and establish ecologically functional habitats will not be achieved. Conservation markets that consider connectivity lead to considerably better conservation results than markets without spatial incentives (Hartig and Drechsler,

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rehabilitation locations and buyers among developers. If the analysis is to be complete, a share of the transaction costs should be charged to each off-set trade (T) for setting up the regulatory conditions for habitat banking market (TCm). We must subtract the benefits (cost savings) of the trade which are the difference in the opportunity costs (OC) between the rehabilitation location (b) with lower opportunity costs and the development location (a) with higher costs.

$$(Ab*Qb + Aa*Qa) / [(TCa - TCb - TCi - TCm/T) - (OCb - OCa)] \quad \text{Eq. 1}$$

2009). Through accumulation of off-sets habitat banking can then target these at connecting habitat and increasing the scale of conservation efforts. While this is a prerequisite of effectiveness for habitat banking, targeting is not a feature unique to habitat banking. Other instruments such as PES can be targeted at priority areas, and connectivity encouraged through the use of agglomeration bonuses.

Trade-off between area and quality equivalence, complex assessment and transaction costs. There is also a trade-off between quality and quantity measures of equivalence. “Complex trading schemes, with individual assessment of rehabilitation sites were found to have substantially lower numbers of transactions, programme participation and hence conservation effect” (Santos et al., 2011a:76). The conservation effect in this quote is measured in terms of number of trades, and presumably area of habitat restoration. There is no evidence as of yet to assess whether the complex trading schemes, through more costly individual evaluation, achieve higher quality of habitat restoration.

Available land, extent of market, equivalence and erosion of conservation targets. Some habitat types and potential restoration locations may become so scarce due to development - e.g. coastal wetlands - that like-for-like off-sets cannot be supplied. Habitats are either developed or already protected by conservation easements. In order to allow further development of coastal wetland sites, alternative off-set sites need to be found in other environments (e.g. inland wetlands). This may lead to pressure from developers and brokers to shift an off-set system from a ‘like-for-like’ habitat equivalence, to off-setting an increasingly limited set of identifiable ecosystem services in increasingly more distant and ecologically dissimilar habitats (Text Box III.3). Scarcity in available off-set sites is seen as a real possibility within the EU (EFTEC et al., 2010). High offset ratios implemented over several decades in a specific landscape are needed to avoid net losses of biodiversity in specific landscapes under a habitat banking scheme (EFTEC et al., 2010). However, limited available rehabilitation sites limit the possibility to use higher compensation ratios to address just such risks and erosion of like-for-like habitat equivalence. Furthermore, a decadal planning horizon would require a strong involvement of a regulator in how and where habitat banks allocated offsets.

Text Box III.3 Equivalence - like-for-like habitat and species-for-species offsets

US regulation on wetland and conservation banking provide an illustration of how equivalence can be implemented in a banking scheme. Equivalence in *wetlands banking* concerns what metric should be used to measure the loss and gain to wetlands or listed species. The federal guidelines suggest that the metric used to assess credits and debits should measure both wetland acreage and function. The credit system should always be expressed and measured in the same way as the impacts of the development projects. Acreage serves as the most commonly used metric used for most wetland mitigation banks even though federal guidance recommends using measures of wetland function instead. Techniques to assess wetland function and to compare habitat suitability include the hydro-geomorphic approach, the Wetland Evaluation Technique, the Wetlands Rapid Assessment Procedure, and Habitat Evaluation Procedures Regulations for wetland mitigation banks generally suggest a 1:1 loss to gain ratio to support the goal of “no overall net loss” of wetland value or function. The goal of the *conservation banking* is to offset the impacts to listed species; therefore, mitigation efforts can be on-site or off-site, depending on whether the affected species or critical habitat is endemic. Federal guidelines require impacts to a particular species or habitat to be compensated for by offsetting losses to the same species or habitat type (e.g., a “species for species” trade-off). (Source: EFTEC 2010:97-98).

(ii) **Opportunity costs**

Without high value under pressure of development and low value land available for rehabilitation and conservation there is no cost-saving to be made by developers nor rent to be made by private brokers of trading and banking, and no additional income to landowners at conservation sites.

Cost and quality differentials. Transfer ratios and conversion factors reflecting differences in land area and habitat quality can make off-set prices/ha inadequate to compensate landowners carrying out rehabilitation, and too expensive for developers wishing to buy credits.

Correlation of habitat equivalence, opportunity costs and extent of individual markets. Allowing trade between different habitat types decreases costs of off-sets, but also reduces effectiveness / the likelihood of finding equivalent habitat. The stricter the like-for-like habitat is practiced, the smaller the area within which trade can take place, because habitat types are expected to be spatially auto-correlated at local scales. Property values and opportunity costs in rural areas are similarly spatially auto-correlated, due to similar biophysical characteristics determining both agricultural productivity and habitat types.

Opportunity cost differential, transaction costs and market size. Off-set credit suppliers face set-up costs, and authorities face monitoring and enforcement costs. These costs need to be covered by a sufficiently large differential in opportunity costs between trading locations for a habitat banking market to be financially viable. Fixed set-up costs also need to be spread across as many trades as possible. A regional sized market with many potential buyers and sellers, spread across locations with large opportunity cost differentials is ideal from a market point of view. In areas where habitat types and environmental conditions are spatially auto-correlated, the need for a large market will be in tension with habitat like-for-like equivalence requirements. For this reason, Brazils TDR system first requires trades to be sought within local catchment of 3^{rd} or 4^{th} tributaries (EFTEC et al., 2010, appendix).

Additionality and opportunity costs. Additionality requirements and opportunities for cost-savings on trading are in conflict. Sites with low opportunity costs are more unlikely to be developed anyway, so there is an incentive for landowners to offer these areas first for a biodiversity off-sets program. When these would have regenerated naturally through lack of use, there is a question of whether the off-sets generated offer benefits that are additional to those that would have come about without the trading scheme. The higher the opportunity costs at the rehabilitation site, the more likely it is that off-sets will be additional, but the lower is the differential in opportunity costs between sites. A low cost saving potential makes the rents that can be made from a trade lower and the market smaller.

(iii) Motivational aspects

Private actors, whether landowners or brokers, seek to minimize costs of complying with development caps and conservation targets. Private motivations to provide public goods beyond regulated caps are exceptional.

Motivations to substitute between compensation possibilities. Like-for-like habitat offsets are costly. If there is to be sufficient demand generated by an environmental liability of a cap on development or a no net loss policy target, developers must have few alternative ways of gaining additional development potential other than biodiversity off-sets.

Biodiversity off-set residuals versus conservation target achievement. Full rehabilitation of a habitat is expected to be more time consuming and costly the more biodiverse and ecologically complex a habitat is. There is therefore a private motivation to rehabilitate quickly and simply. There is evidence that rehabilitation projects have fallen short of re-establishing reference/base-line levels of biodiversity and ecosystem service provision (Benayas *et al.*, 2009). This leads to biodiversity off-set residuals also at the offset site (location B, in the figure III.2 above).

'Licence to trash' versus 'trading up'. Legal compensation possibilities may open up for local government approval of destructive development options, motivated by local interests to capture rents from development. On the other hand, companies and municipalities wishing to increase public goodwill as part of e.g. corporate social responsibility agenda, may be motivated to buy extra or higher quality or offsets in what is known as 'trading-up'.

(iv) Transaction costs

The smaller the opportunity cost differentials between development and off-set sites, the more limited the available habitat sites, the more options there are to avoid complying with equivalence, and the higher the uncertainty about results, the more need there is for a public-interest regulator. Transaction costs of implementing a habitat banking scheme will increase correspondingly.

Equivalency and transaction costs. Achieving equivalency between the type of damage and the off-sets is costly. Monitoring and evaluation of on-site mitigation measures and off-site off-setting measures implies costs additional to monitoring of compliance of land use regulation (without trading).

Time equivalency, transaction costs and extent of market. In habitat banking a rapid loss due to development is traded for a slower gain in the rehabilitated habitat. In the period from development to full rehabilitation in an offset site, damage may go uncompensated, requiring additional off-set area to compensate for interim damages. Long rehabilitation times create uncertainty regarding effectiveness, requiring either an additional area of off-sets to compensate for the risk of not achieving full rehabilitation in the long term, or limiting the habitat banking market to only habitats that have rapid restoration times (e.g. wetland creation which measured in terms of area is rapid, versus forest regeneration which is slow). Rapid restoration habitats are by definition less of a biodiversity conservation issue, than slow generation habitats. The habitat banking market would therefore be limited to sites of less conservation interest for biodiversity, even though they may still be of interest to a particular species or ecosystem service (e.g. fish and recreation in the case of wetlands).

Information costs vs. opportunity costs. Where trading takes place in a similar environment, say within a catchment, the information costs are expected to be lower for locating land with similar habitat to that lost to development. However, opportunity cost differentials between locations can also be expected to be low. In larger markets offering higher opportunity cost differentials, the information costs of identifying sellers and buyers of off-sets may also be higher, requiring some type of information broker such as a habitat bank. The increased information costs of the broker are recovered from the opportunity cost savings on each trade through a surcharge.

Private internalization of environmental costs, through externalization of transaction costs to the public. The larger the off-set market the more likely it is that a habitat bank can recover its brokering costs, that landowner's opportunity costs of generating off-sets can be compensated, and that developers can reduce their costs of impact mitigation. However, we argued above that this has the potential to lead to greater information costs in finding off-set habitats that are equivalent. Monitoring and enforcement costs are required at both rehabilitating and developing locations – is there reason to expect that these costs would be lower than under land use zoning schemes without off-set trading? Monitoring and enforcement also need to be carried out by a third party, which makes it likely that a public regulator must carry some of the costs.

Land-use zoning, alternative, baseline or precondition? Habitat banking is sometimes compared to "a command-and control system of zoning alone" (Santos et al., 2011a:75). Habitat banking has the potential to let developers internalize the impacts of their development on habitats at a lower cost, relative to a pre-existing land use zoning scheme without trading. However, it is not clear that habitat banking reduces total transaction costs for society, given that land-use zoning and development caps are a precondition for establishing an off-set market in the first place, rather than an alternative to it.

2.4.2 Equity

Equity concerns the distributional effects of the chosen system – who suffers from losses and who benefits from gains of habitat banking? Reviews of habitat banking have placed less emphasis on this aspect than effectiveness and cost issues. Santos et al. (2011a) address several points, including the prerequisite stakeholder involvement, increasing social justice of zoning regulations, the size of the market area, new source of income for local communities, receiving area equity and tenure equity for landowners.

Most authors agree that involvement of stakeholders at both the regenerating and receiving site for off-sets is a prerequisite for the legitimacy of changing land-use practices. There is also a broad perception that TDR schemes increase the social justice of zoning regulations because development restrictions may be compensated in zones generating off-sets, while developers have to pay for additional development exceeding the prior legal limit development cap (Santos et al., 2011a). However, this assumes that the prior existing zoning scheme and the development restrictions were both ineffective and unjust.

Development of site ecosystem services and size of market. As seen above, cost-minimization of a habitat banking scheme is expected to be greater in a larger market, while the effectiveness of larger areas is in doubt because of increasing problems of finding like-for-like habitats. Credit sites far away from the damage may also not compensate stakeholders around the development site for adverse effects (EFTEC et al., 2010). This critique recognizes that the habitat lost to development may have been generating ecosystem services to the local population, which are not accounted for in the financial compensation to the landowner herself. Some TDR schemes in

Brazil try to minimize this possible ‘externality’ on neighbors by prioritizing trades within the same catchment.

New sources of income for local communities. Transaction of development or habitat credits is generally regarded as a new source of income for local communities generating the credits and those involved in the habitat banking system. A closer look at expected employment generation from habitat banking schemes reveals that employment opportunities are composed almost entirely of trained professionals who would not be expected to live in rural communities generating the off-sets (Table III.3). If anything, this preliminary analysis of employment opportunities shows that development areas and urban communities offering financial and technical services would be the ones to benefit from habitat banking schemes. The employment opportunities listed in Table III.3 also provide an indication of the transaction costs involved in running a habitat banking scheme.

Tenure equity for landowners. Compensation of landowners for opportunity costs of providing additional habitat for off-sets has been criticized for applying flat rates that do not provide a premium for ecologically critical parcels, agglomeration of rehabilitated sites, or land under high development pressure. There are some problems with these arguments. While land under high development pressure would be expected to fetch high market prices this may not be captured if residents don’t have full land rights. On the other hand, ecologically critical parcels would not be expected to be open for trading if they could be identified as critical *ex ante*. Agglomeration bonuses could be recognized by differentiating off-set types. Other authors question the fact that habitat banking – as with PES in general – seeks ‘lowest cost of conservation’ opportunities, compensating landowners at their current level of poverty (Karsenty, 2007). Especially in developing countries conservation easements would lock local communities out from any development opportunities and future gains in the value of their land. This is obviously a ‘hard’ trade-off of conservation versus development faced by all compensation-based conservation instruments.

Table III.3 Distribution of employment opportunities with habitat banking Source: Bovarnick et al., (2010)

Employment type	Specific employment opportunities
Design, establishment and maintenance of habitat banks	Wetland conservation scientists, biodiversity conservation scientists, hydrological engineers, conservation wardens, landscape engineers, forestry professionals, habitat restoration experts, construction workers
Monitoring, evaluation and verification	Wetland conservation scientists, biodiversity conservation scientists, forestry professionals, habitat restoration experts
Legal support	Property lawyers, financial lawyers
Registry and administration	Market administrators, registry specialists, public administrators
Project finance & banking services	Investment bankers, venture capitalists, commercial bankers
Market information services	Market researchers, news and intelligence analysts
Fund creation and management	Investment fund managers, fund management consultants
Project technical support	Environmental consultants with knowledge of habitat and wetland restoration, NGO specialists, researchers

Receiving area resident outcome equity. Agglomeration of development in areas buying biodiversity offsets or TDRs can also lead to environmental impacts that are not accounted for in the calculations of habitat compensation requirements. For example, development may lead to congestion and pollution issues at the development site. Lower accessibility of natural environments for local communities may be another effect of 'running down' a cap on local development. However, these are not meant to be dealt with by biodiversity off-sets, but through the strategic environmental assessment and implementation of the mitigation hierarchy on-site by the developer.

Uncertainty, risk and equity. Predicting mitigation needs on and off-site is more complicated than Figures III.1-2 suggest due to variability in natural processes, in development pressures on land use, and time-lags in rehabilitation of habitats. Uncertainty makes predicting 'net habitat' at sites A&B more difficult, and differences between ecosystem services lost/gained between sites more unpredictable (Figure III.4). Time lags in rehabilitation – especially for more complex habitats – increase this uncertainty, risk of incomplete rehabilitation and failure to offset. Increasing off-set ratios or development of secondary insurance and bond markets may 'offset the risk of offsets', but this in turn raises the transaction costs and questions about equity (these legitimacy issues are discussed further in Appendix).

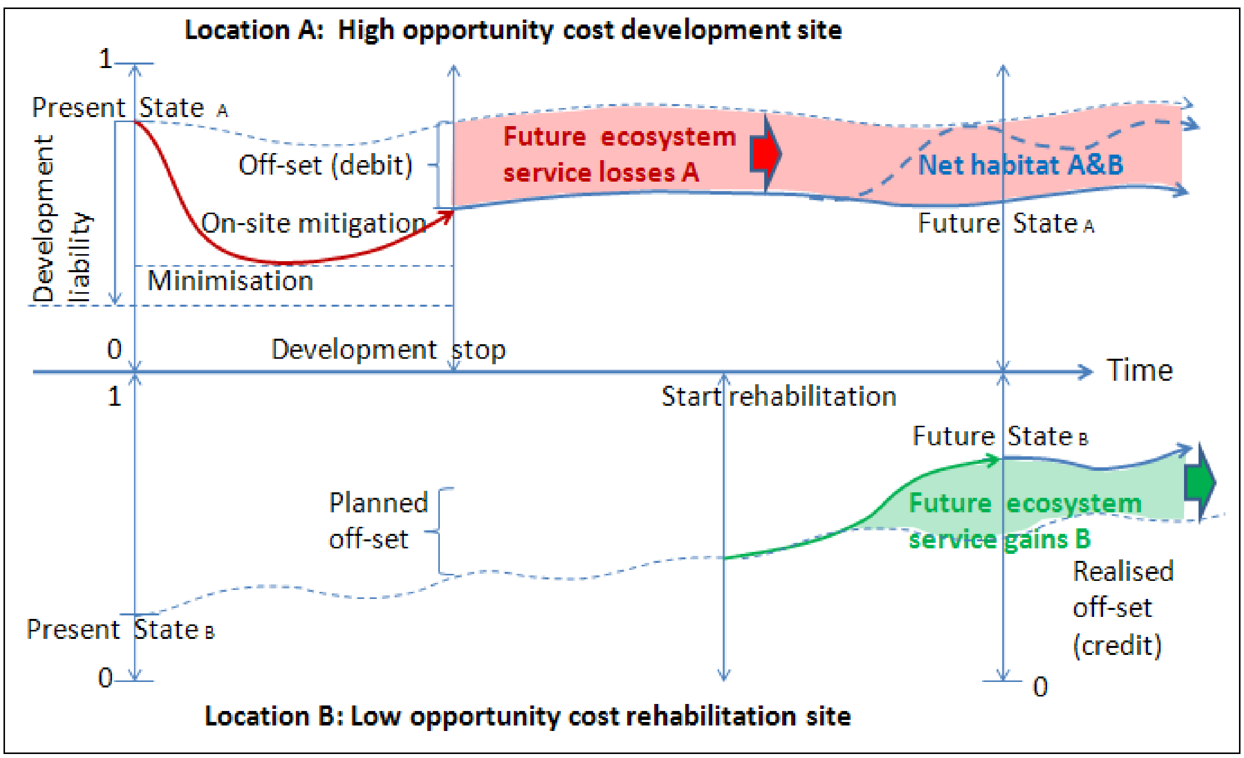

Figure III.4. Equivalence of ecosystem services gains and losses with ecosystem dynamics

3. ECOLOGICAL FISCAL TRANSFERS[34]

3.1 DEFINING ECOLOGICAL FISCAL TRANSFERS IN BIODIVERSITY CONSERVATION

Protected area regulations as typical regulatory instruments on their own are not enough. Instead, a combination of regulation and economic instruments capable of offsetting the costs associated with protected areas is required. Whereas PES schemes reviewed in Part II address payments to private actors (often with public funding), ecological fiscal transfers are transfers of public funds from central to local government level to compensate for opportunity costs to local government of establishing and managing conservation areas or other environmental measures. They may also be motivated by positive spill-over benefits to nearby areas or the country at large.

Intergovernmental fiscal transfers help lower level governments cover expenditures in providing public goods and services. Fiscal transfers from central to local level very often have a 'fiscal equalization' purpose to adjust for local government fiscal capacity and needs so that public service levels per capita at the local level are more equal. Fiscal transfers may also be earmarked to the implementation of certain central policies, and conditional on local government performance. However, the bulk of fiscal transfers is allocated in the form of lump-sum or general purpose (unconditional) transfers.

Decisions about where conservation areas are to be sited are frequently taken at higher levels of government, even though the costs of losing those areas for other social and income-generating developments are borne by local governments and communities. Ecological fiscal transfers are therefore seen as an instrument to provide incentives for local governments to support and maintain the quality of water and nature conservation areas within their territories, but which can also provide wider ecological benefits beyond municipal boundaries (Ring, 2008; TEEB, 2009).

Ecological fiscal transfers for biodiversity conservation build on existing protected area regulation in that they use officially designated protected areas – their area and sometimes also conservation-based indicators of quality – as an indicator to allocate fiscal transfers. Ecological fiscal transfers have the potential to turn the oft-encountered local opposition towards protected areas into active support, but for this to occur requires that municipalities and/or state governments inform the citizenry and local officials of the relation between protected areas and the additional revenues, and make an effort to reward them for enhancement in biodiversity protection in an adaptive governance strategy.

The funding involved in fiscal transfers may have as its source tax revenue or redistribution of international transfers of funds such as REDD+. By addressing local government land-use decisions, ecological fiscal transfers complement a policymix of economic and regulatory instruments largely addressing private actors.

[34] The text of this chapter is a summary of a review of ecological fiscal transfers conducted by Irene Ring and colleagues for the EU FP7 funded POLICYMIX project (Ring *et al.*, 2011).

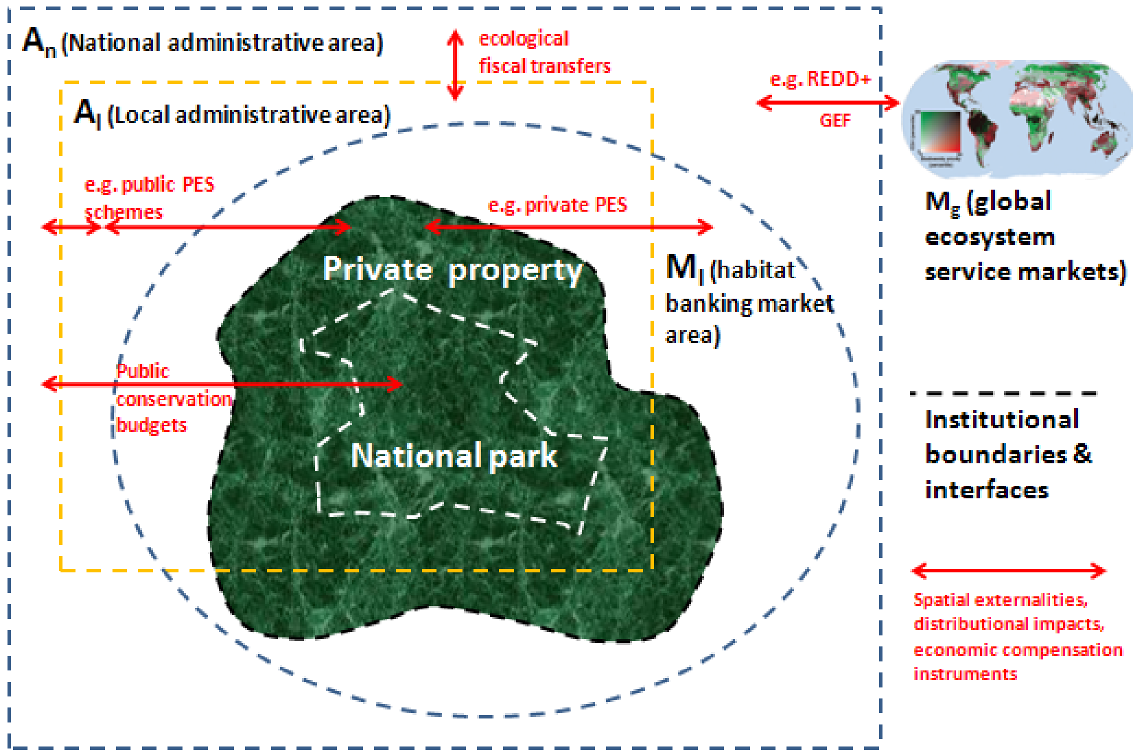

Figure III.5 Ecological fiscal transfers complementing other economic instruments in a policy mix

Ecological fiscal transfers are transfers of public funds from a central government to a local government level to compensate for opportunity costs to local government of conservation areas or other environmental measures. They may also be motivated by positive spill-over benefits to other governments that may range in the case of biodiversity conservation from local to national and even global benefits. Public funding may have as its source tax revenue or redistribution of international transfers of funds such as REDD+. By addressing local government land-use decisions, ecological fiscal transfers complement a policy mix of other economic and regulatory instruments largely addressing private actors.

How widespread are ecological fiscal transfers? So far, fiscal transfers with biodiversity conservation-related objectives have only been implemented in a couple of countries and are at present a rather minor part of intergovernmental fiscal transfer schemes. Only Brazil and, more recently, Portugal have implemented fiscal transfers for biodiversity conservation. Starting with Paraná state in 1991, Brazil became the first country to introduce their ICMS Ecológico – financed through a percentage reallocation of value-added taxes collected at the state level. It now exists in a total of 13 Brazilian states to compensate municipalities for land- use restrictions imposed by protected areas (Figure III.6).

Figure III.6 implementation of ICMS-Ecológico in Brazil (2008)

ICMS-Ecológico (ICMS-E) is an ecological fiscal transfer using ecological indicators to redistribute state-level value-added tax on goods and services back to local level. By 2008, 12 Brazilian states had implemented ICMS-E (map). By 2011, 15 states have legislation and 13 states have implemented the scheme (Ring et al., 2011). (Map Source: Ring et al., 2010)

Starting with experiences from Brazil, ecological fiscal transfers have been increasingly studied during the past decade (Grieg-Gran, 2000; Loureiro, 2002; May *et al.*, 2002; Ring, 2008). In a number of other countries, ecological fiscal transfers for biodiversity conservation have so far only been proposed, and potential consequences partly modeled including Germany, Indonesia and India (Ring et al., 2011).

Within the European Union, Portugal is the pioneer in introducing ecological fiscal transfers. Portugal is the first EU Member State to recognize protected areas as an indicator for the redistribution of public revenues through fiscal transfers from the national to local governmental level (Santos *et al.*, 2010). Portugal's legislation on ecological fiscal transfers was introduced in 2007 and is so recent that it is too early to observe any change in protected area coverage and local public agents' decisions and attitudes towards conservation.

In Norwegian conservation policy, the new Nature Diversity Law of 2009 mandates state compensation for costs of establishing land-use regulations for biodiversity conservation (NOU, 2009)[35]. In the near future, ecological fiscal transfers may also play a role in the implementation of international programs on a nationwide scale, linking climate mitigation with biodiversity conservation policies (Ring *et al.*, 2010). For example, REDD+ initiatives (Angelsen 2008) actively supported by the Norwegian Government, will need to take into account fiscal transfer schemes to the local level as one important means of channeling international payments for biodiversity conservation and climate mitigation from the national down to lower levels of government (Ring et al., 2010).

[35] In this regard, Friends of the Earth Norway have proposed that the fiscal transfer formula (the "kommunenøkkel" – or municipal key) be updated "so that those municipalities that are best at preserving nature and thereby biodiversity and ecosystem services are rewarded" http://naturvernforbundet.no/nyheter/loennsomhet-i-naturbevaring-article17382-166.html and Kommunal Rapport 10.9.10.

3.2 POLICY AND DESIGN ISSUES

Existing ecological fiscal transfers in Brazilian states and Portugal use officially designated protected areas and/or areas designated as water catchments for drinking water provision as indicators to allocate lump-sum transfers to local governments.

In most schemes, just the protected area coverage as a quantitative indicator is used. In Portugal, transfers per hectare protected area are higher, if protected area coverage in relation to municipal area is beyond 70 % (Santos et al., 2010). In Brazil, the different categories of protected areas are further multiplied by a conservation factor or weight, reflecting the varying land-use restrictions associated with, for example, strictly protected or sustainable use areas (Grieg-Gran, 2000; May *et al.*, 2002; João, 2004; Ring, 2008). For example, parks, reserves and ecological stations are afforded the highest weight (0.7-1.0), environmental protection areas (APAs) which offer many options for direct use are given the lowest weight (0.1), whereas private nature reserves (RPPNs) range in between (0.2-1.0) (Ring et al., 2011). States such as Paraná and Minas Gerais, have introduced a quality indicator of protection into relevant legislation. The qualitative aspect of the Paraná allocation scheme is unique, referring to aspects which are judged to improve the relative degree of conservation integrity of those areas protected within a given municipality.

3.3 OUTCOME LEGITIMACY

3.3.1 Effectiveness

Environmental effectiveness of ecological fiscal transfers has not been explicitly addressed in the literature. Being a compensatory measure, there is no additional environmental objective to be achieved with the policy instrument. The baseline for ecological fiscal transfers for biodiversity conservation may be interpreted as the amount of designated protected area when the instrument is introduced, though the lump-sum payment related to ecological fiscal transfers is adjusted each year in recognition of additional protected areas that come into being.

In principle, recognizing the positive spatial externalities associated with protected areas provides a positive incentive for municipalities to acknowledge and value the natural capital within municipal boundaries, when this is otherwise mostly perceived as an obstacle to development. No recognition of positive externalities – whether they are provided by private or municipal actors – is expected to lead to an under-provision of the relevant public goods and services, in this case nature conservation (Ring, 2008a).

Although the ICMS Ecológico (ICMS-E) has originally been introduced as a compensation for land-use restrictions, it developed at the same time as an incentive to create new protected areas (May et al., 2002). The increase in protected areas in a number of Brazilian states after the introduction of ecological fiscal transfers has been attributed to ICMS-E (Bernardes, 1999, Loureiro, 2002, May et al., 2002, Ring 2008a). Recent numbers for Paraná indicate that in total, protected areas have increased by 164.5 % since the introduction of the ICMS-E (Ring et al., 2011). This increase has mostly taken place within the first 10 years after the instrument had been introduced, indicating a possible saturation effect as low opportunity cost and available land for protected areas becomes scarce. In order to establish attribution of ICMS-E to protected area creation it is also important to check whether the baseline area of protected areas was static or in an upward trend at the time of the instrument's introduction in the relevant state. For ICMS-E to have an incentive effect indicators in these schemes need to be sufficiently easy to grasp and monitor, and statistically available (Ring, 2008a).

In Portugal, the analysis presented by Santos et al. (2010) shows that ecological fiscal transfers can be significant for some municipalities in which the amount of land granted conservation status constitutes a large part of their overall territory. The ecological criterion may thus work as an incentive for municipalities to maintain or increase their protected areas; in terms of quantity, however, this must be complemented with quality criteria to also provide incentives for the management of those areas. As the Portuguese scheme has only been introduced in 2007, it is too early to identify an incentive effect in practice. The new Law by simultaneously introducing a significant number of changes in the Portuguese intergovernmental fiscal transfer scheme, makes the ecological component difficult to grasp by the stakeholders affected (namely municipal authorities) due to the presence of many crossover effects (Santos *et al.*, 2011b). Therefore, informative instruments and activities are an important factor to motivate municipalities for more and better biodiversity conservation.

3.3.2 Efficiency

Introduction costs for ecological fiscal transfers are reasonably low. This holds especially true if easily available indicators are used, such as the protected area coverage (Ring, 2008a). Costs of conservation measures themselves do not apply to ecological fiscal transfers if the aim of the instrument is only to compensate for opportunity costs. The implementation costs of ecological fiscal transfers are comparatively low because they do not require new institutions or a new bureaucracy (Ring, 2008a). By introducing an easily available ecological indicator into the existing fiscal transfer mechanism such as protected area size, ecological fiscal transfers build on existing institutions and administrative procedures. This may not be true, however, for the implementation of qualitative indicators, which requires a regular field validation of protected area management quality and relevance to local sustainable development. If a quality criterion is implemented, like in the case of Paraná, the quality of protected areas needs to be monitored at regular intervals by conservation authorities. However, the effectiveness of ecological fiscal transfers for biodiversity conservation is far greater with the implementation of these parameters and some kind of conservation monitoring related to performance of protected areas is often required anyway (e.g., Natura 2000 network in Europe). Therefore, quality monitoring for ecological fiscal transfers should build on regular conservation monitoring activities executed by conservation authorities to increase efficiency.

3.3.3 Equity

The social impacts of ecological fiscal transfers are related to redistribution effects of resources between municipalities, any additional impacts caused by municipal spending, and the effects of formal protected areas on local land users.

In Latin American countries, most public revenues still stem from value-added taxes that hit the poor harder than the rich. To the extent that ecological fiscal transfers build on public revenue from value-added taxes the effects will be regressive. Currently, the schemes in operation in Brazil only redistribute a portion of the existing VAT. More generally, distributive aspects need primarily to be dealt with as part of the general tax system of a country.

In the case of the ICMS Ecológico in Brazil, a certain percentage of state ICMS revenues – the most important tax in terms of public revenues at the state level – is reserved for distribution among local governments with conservation units. This clearly leads to winning and losing municipalities because other indicators have been lowered with the introduction of the ICMS Ecológico. Nevertheless, the success of ecological fiscal transfers is linked to a strong tax, the revenues of which usually show an increasing trend over the years, especially in times of economic growth.

In other countries, ecological indicators may also be introduced as part of the fiscal need determination of a jurisdiction, which is then entered in a formula-based procedure and weighted against the fiscal capacity of the relevant jurisdiction. If own source revenues are low and protected area coverage high, then the new ecological indicator increases fiscal need and thus the transfers received by the municipality. In this way, poorer municipalities in rural areas with low revenues (e.g. due to low population densities and/or little economic activities) and high protected area coverage benefit most (Ring, 2008b).

At present none of the ecological fiscal transfer schemes in operation in Brazilian states or in Portugal allow earmarking of the transfer on conservation-related spending in municipalities. They have been introduced using funds where constitutional regulations foresee lump-sum transfers to guarantee municipal financial autonomy. This also means that the new transfers could be used for environmentally harmful activities, even to the point of promoting activities that potentially destroy or degrade valuable habitats. Therefore, it is important to consider both quantity- and quality-related indicators in ecological fiscal transfers.

To the extent that ecological fiscal transfers work as an incentive for park creation at the local level, the instrument also has some social impacts on landowners within the parks. 'Effective management' of parks and reserves that are meant for exclusion of humans may mean clamping down on social benefits derived from low impact uses of such lands by communities neighboring on or residing within these areas. Direct compensation of landowners for e.g. expropriation or use restrictions of national park creation is not addressed with ecological fiscal transfers. Because municipalities have financial autonomy, and the transfers are mainly directed at compensating for financial costs (e.g. loss of tax revenues) from avoided development, ecological fiscal transfers are not generally used to compensate landowners. However, there is evidence in Brazil that municipalities have used funds to encourage private conservation efforts (Ring, 2008).

4. REFORMING SUBSIDIES

This section gives a brief assessment of subsidy reform as a potentially important component of any mix of instruments to increase potential financing for conservation and create more appropriate incentives. Subsidy reform is not a new mechanism as such, but has been placed in Part III of this report because reform processes are in their infancy in most parts of the world and substantial progress is necessary. The TEEB study, for example, calls for 'doubled efforts' to reform subsidies (ten Brink et al., 2009).

4.1 WHAT ARE SUBSIDIES?

Subsidies come in many shapes and forms. A common definition of a subsidy is "….government action that confers an advantage on consumers or producers in order to supplement their income or lower their cost" (OECD, 2005). Subsidies are most commonly thought of as direct transfers (or potential transfers, e.g., covering liabilities) from the government to private or civil-society actors. But they may also consist of income or price support, tax credits, exemptions and rebates, loans on special terms, and preferential treatment of various kinds (ten Brink et al, 2009).

Some of the subsidies can be read out of public budgets (e.g. national accounts) and some are not-accounted for ('off-budget'). Due to the wide heterogeneity of subsidy types and varying degree of formal accounting, statistics on subsidies are fairly sketchy. However, work is ongoing to systematically record and calculate direct and indirect subsidies, for example in the World Trade Organization.

According to the definition above, and as discussed in Part II of this report, PES schemes financed by governments are subsidies. Subsidies are introduced and maintained for various social, environmental and economic reasons. Many of these are good or politically rational reasons (ten Brink 2009). For example, as emphasized in environmental economics, a subsidy can be sensible and efficient if given to reduce so-called environmental externalities or encourage positive ones. PES schemes try to stimulate the generation of positive externalities (or avoidance of further negative ones), given that landowners often in practice are seen to have the right to keeping the existing situation (or to cause further environmental degradation – as discussed in Part I). While renaming 'subsidies' to 'payments' may seem to have increased their legitimacy, caution is necessary. As pointed out by ten Brink et al (2009) even so-called green subsidies may distort economies and markets, and may not be well-targeted or cost-effective – as we have also seen with several ongoing and previous PES schemes.

Many subsidies all too often end up distorting prices and resource allocation decisions, typically also harming the environment. Typical examples of such subsidies are the ones stimulating over-production (e.g. within agriculture) or over-consumption (e.g. the use of coal in energy production, petrol for cars). Whether subsidies are good or bad often comes down to their objective and their specific design and implementation. Pieters (2003) include the following questions:

- Do they serve (or continue to serve) their intended purpose?
- At what cost (efficiency)?
- How are the costs and benefits distributed (equity)?

- Are they harmful to the environment in general and for ecosystem services and biodiversity in particular?

4.2 REFORMING SUBSIDIES – A POTENTIAL WIN-WIN?

The overall level of subsidies is enormous. Subsidies to agriculture, the largest of all subsidies, are estimated at 250 billion US$ per year in OECD countries alone (ten Brink, 2009). Subsidy reform has been on many government agendas since the early 1990s and has been pushed after 2000 by, for example, a series of studies by the OECD (OECD, 2003, 2005).

The main rationale for subsidy reform is to restore resource allocation efficiency, i.e. direct scarce resources to areas of production and consumption where they are valued the highest. Further, subsidies are typically financed by taxes that create their own efficiency losses in collection. Finally, as inefficient subsidies are cut, scarce government resources can be freed for more productive uses, including funding of conservation policies. Although some subsidies may be particularly bad for biodiversity and ecosystem services, e.g., biofuel subsidies and agricultural subsidies that stimulate over-production in general (OECD, 2008), it is important not to restrict subsidy reform only to those subsidies that are explicitly bad for the environment.

Achieving these efficiency gains through subsidy reform (or removal) has proven tremendously challenging. Although the *effectiveness* and *efficiency* arguments for removing or reducing many types of subsidies are compelling, issues of *equity and process legitimacy*, and share complexity of reform, are effectively halting progress in this area.

As concluded by Pearce and von Finckenstein (2000):

"The complexity arises from the fact that subsidies are manifestations of rent-seeking, which, in turn, is part of a wider category of unproductive activity in economic systems. Rent-seeking involves redirecting economic resources to special interest groups rather than using resources productively. Interest groups then use those resources to reinforce their privileged positions. Subsidy reform will inevitably conflict with those special interests. The idea that subsidy reform is a "win-win" policy is therefore misleading – there will always be losers, even if they are undeserving losers. In many cases, the most harmful subsidies will be those that are least easy to remove".

This quote may be a bit strong, as some subsidies may have legitimate and sensible objectives. However, a large share of subsidies seems still to fall in the category criticized by Pearce and von Finkenstein.

Hence, removing or reducing ineffective subsidies that have no current legitimate objective is often a painful process for the interest groups that stand to lose and they will fiercely oppose such changes. Such potential conflicts can be sought alleviated through broad stakeholder engagement, transitional assistance (to ease the pain), increased transparency and information exchange (which may increase the broader support for reform). Ten Brink et al. (2009) have drawn up a road map for reform of subsidies (see Box III.4 below), acknowledging the equity issues involved in subsidy reform.

Text Box III.4 Developing a road map for reform: a checklist for policy-makers

14. Assess the costs and benefits of potential reform in more detail:

- potential **environmental benefits**: include thinking on benefits in other countries and secondary effects, which can be perverse;
- potential **economic costs**: e.g. national (tax, GDP, etc), sector-wide, for winners and losers within the sector (including new entrants/future industry), for consumers/citizens (affordability);
- potential **social impacts**: e.g. jobs, skills, availability of goods/services, health;
- potential **competitiveness and innovation benefits**
- potential **ethical benefits** e.g. as regard fairness of income, appropriateness of support, links to future generations;
- is the reform **practical and enforceable**?

To identify the likelihood of success and whether it is worthwhile using political capital for reform, the following questions can be useful to set priorities for the road map.

Is there a policy/political opportunity for action?

15. Is there a window of opportunity? e.g. policy review process, evaluation, public demand?
16. Is there a potential policy champion?
17. Will there be sufficient political capital for success?

These questions can be answered at different levels. A quick scan can help develop the overall picture, but more detailed analysis is needed to clarify the details, identify what should be the exact nature of the reform and support the call for subsidy reform.

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Source: ten Brink et al., (2009)

4.3 CONCLUSIONS AND TRANSFERABILITY TO DEVELOPING COUNTRY CONTEXTS

As we have seen, some subsidies may serve sensible and legitimate purposes such as many PES schemes. However, a larger share of subsidies creates distortions, demand scarce public resources and may be harmful to the environment. In a developing country context, such subsidies are often closely linked to social policies aimed to alleviate poverty. However, they are also often linked to corruption, making change difficult. Reform of subsidies should in developing countries go together with broader macroeconomic and political reforms. However, as pointed out by Pearce and Finckenstein (2000), democratic political reforms are not sufficient as western democracies have higher subsidies than the rest of the world.

Interestingly, in these days of government fiscal crises, Pearce and Finckenstein (2000) argue that moments of crisis may be transformational, i.e. give governments a ‘sudden shock’ pushing reform:

> *“Some have advocated “sudden shocks” whereby dramatic events are seized as an opportunity to institute reform. There is some evidence to suggest that if a crisis does occur, it may be best to implement subsidy reform along with other transitional measures in one large package An alternative is to let the almost inevitable growth of subsidies produce economic bankruptcy, and then institute reform. But many societies have proved surprisingly resilient whilst sustaining extensive subsidy regimes, and the costs of waiting may not be acceptable anyway.”*

It remains to be seen whether the current “fiscal” window of opportunity is seized, potentially freeing financing for other policies, including biodiversity and ecosystem policies.

APPENDIX. HABITAT BANKING – EFFECTIVENESS, EQUITY ANDFAIRNESS IN THE PRESENCE OF HABITAT DYNAMICS

by

David N. Barton

The independence in the timing of assessment of damage (debits) and assessment of offsets (credits) is the key feature distinguishing habitat banking from biodiversity offsets (EFTEC 2010). EFTEC (2010) has a comprehensive discussion of issues that remain to be resolved for habitat banking to become a reality. The report addresses as 'Secondary issues' the topics of managing risk, accounting for time preference, achieving conservation targets and social equity. In this appendix we devote some more attention to these 'secondary issues', believing that uncertainty of habitat and land-use dynamics may make them key to stakeholders' interpretation of the fairness of habitat banking.

Either development takes place at location A before the rehabilitation project at location B, creating an offset debit (Figure A1), or rehabilitation takes place before the development in which case an offset credit is generated (Figure A2), EFTEC(2010).

Particular problems for the legitimacy of habitat banking in separating offsetting activities from development temporally and spatially in the presence of dynamics and uncertainty are explained below with the help of a series of diagrams:

1) compensation of interim habitat losses (Figure A3)
2) compensation of ecosystem service between sites (Figure A3)
3) uncertainty in predicting biodiversity offset and achieving a "no net loss" conservation target in a dynamic ecosystem (Figure A4)
4) uncertainty regarding the distribution of ecosystem service gains and losses between the development site and the offset site (Figure A4)
5) restoration time lags for the most biodiverse and complex ecosystem types, and problems of discounting present and ecosystem service losses against future gains. (Table A1)

1. Compensation of interim habitat losses (Figure A3)

Off-sets are ideally generated ex ante a development and stored as credits until purchased. Given long lead times in identifying restoration sites and time lags in restoration that are longer the more complex the ecosystem (Table A1), there is a risk that off-sets are generated ex post of a development. Assuming here that an equivalent habitat can be found, there are still interim losses in habitat and ecosystem services which would require offsetting. A higher than 1:1 ratio offset would be required to compensate for interim loss.

2. **Compensation of ecosystem service between sites (Figure A3)**

In Figure A3 we assumed that (i) opportunity costs are fully compensated, (ii) that there are no dynamics in the habitat so that the future states of ecosystems at locations B and A are known, (iii) that there is no net loss of habitat by the time rehabilitation is completed. Even under these ideal conditions ecosystem service losses at site A may not have been compensated by ecosystem service gains at site B, because the constellations of users, their locations relative to the development and off-set sites, and their use intensity at locations A and B are likely to be different. While opportunity costs of landowners at site B are compensated, and habitats equivalently restored (their composition, structure and function on-site), loss of ecosystem services at and around site A is not necessarily compensated by gains at and around site B. Examples of such externalities include loss of landscape cultural values (aesthetics, recreational opportunities) for populations around the development site that are not compensated by off-sets at a different site and time unless they are equivalent in terms of access.

3. **Uncertainty in predicting biodiversity offset and achieving a "no net loss" conservation target in a dynamic ecosystem (Figure A4)**

In Figure A4 we introduce some dynamics in terms of habitat quality fluctuating at location A due to a natural variability – this might be the example of habitat extent and structure changing between years due to e.g. droughts. At location B we assume both natural variability and a downward trend in habitat extent/quality due to human pressure. This may be the case where increasing population is expected in an initially rural area that is being used for habitat rehabilitation for habitat banking purposes. Uncertainty in the baseline future states of habitat at both locations make planning for the offsets demanded more difficult. The challenge is illustrated in the need to predict the line in Figure A4 "net habitat A&B". In order to guarantee that equivalent habitat is rehabilitated larger offset ratios than 1:1 are required, the ratio being larger, the greater the variability between the sites. Differences in land-use development trends between sites would increase the ratio further. Increasing offset ratios comes at the cost of rehabilitating additional areas relative to a situation with no uncertainty (as in Figure 3A). This is an illustration of the cost of uncertainty to a habitat banking scheme. Perception of this uncertainty by stakeholders at both sites may magnify calls for increasing the off-set ratios between sites, and if not addressed jeopardize perceived fairness of the habitat banking schemes.

4. **Uncertainty regarding the distribution of ecosystem service gains and losses between the development site and the offset site (Figure A4)**

Differences in natural variability and changing land-use pressures between the sites could also affect the expectations stakeholders have of losses of ecosystem services at site A and gains at site B. This challenge is illustrated by the need to predict – at the time the development is completed and the off-set is demanded – the losses and gains in uncertain ecosystem services as a result of uncertain changes in habitat (the latter represented as red and green areas of Figure A4). Uncertainty would be expected to magnify problems with stakeholders perception of fairness addressed under (3) above.

5. **Restoration time lags and problems of discounting present and ecosystem service losses against future gains (Table 1A and Figure A4)**

Table 1A (EFTEC et al. 2010) shows how restoration times for different habitats and landscape features. A hypothesis is that more biodiverse and complex structured ecosystems have longer restoration times. For these systems there is greater likelihood that rehabilitation is completed and offsets credits verified after the development takes place. This is illustrated in Figure A4. Because interim losses in habitat and ecosystem services (location A) may occur many years in advance of complete rehabilitation at the offset site (location B). Discounting of future benefits against present costs imply that less habitat restoration may be required in future to compensate for a loss now, because the economic value of credits is made equivalent through discounting (EFTEC et al. 2010). Discount rates are also higher for riskier prospects.

Some closing considerations regarding equity and legitimacy

EFTEC et al. (2010) state that an advantage of banking systems is that a precondition can be to establish offset credits before sale takes place. Given the time lags in identifying habitats and restoration itself this requirement would push establishment of habitat banking farther into the future than the 10 years predicted in the report. At the same time financial and development sectors have an economic interest in pushing for earlier implementation, before credits have been generated. While habitat banking systems are developing ways to address discounting and speculation in this risk (EFTEC et al. 2010, case appendix), we think that habitat banking also faces problems in explaining the rationale of discounting and its biophysical consequences for offsets to non-financial and non-development stakeholders. An offsets scheme that delivers economically equivalent, rather than biophysical equivalent, offsets is unlikely to be perceived as neither precautionary nor equitable.

Figure A1. Habitat banking (biodiversity offset debit with ex post rehabilitation)

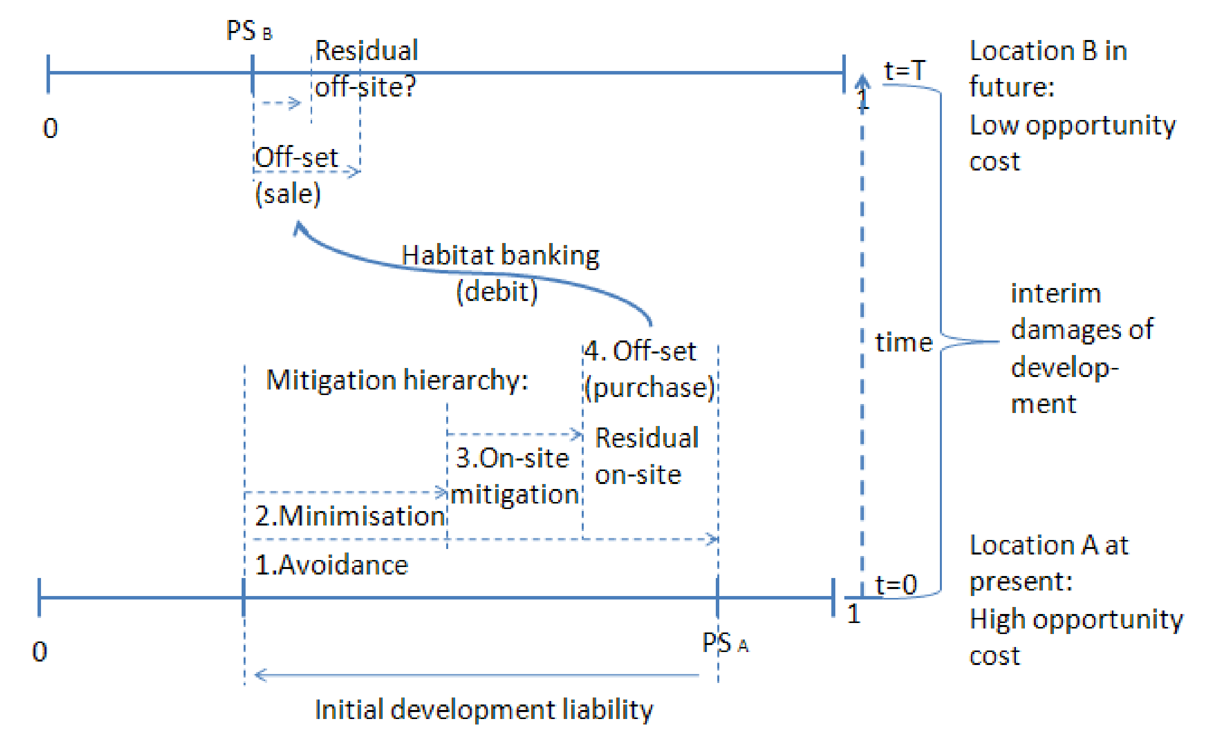

Figure A2. Habitat banking (biodiversity offset credit with ex ante rehabilitation)

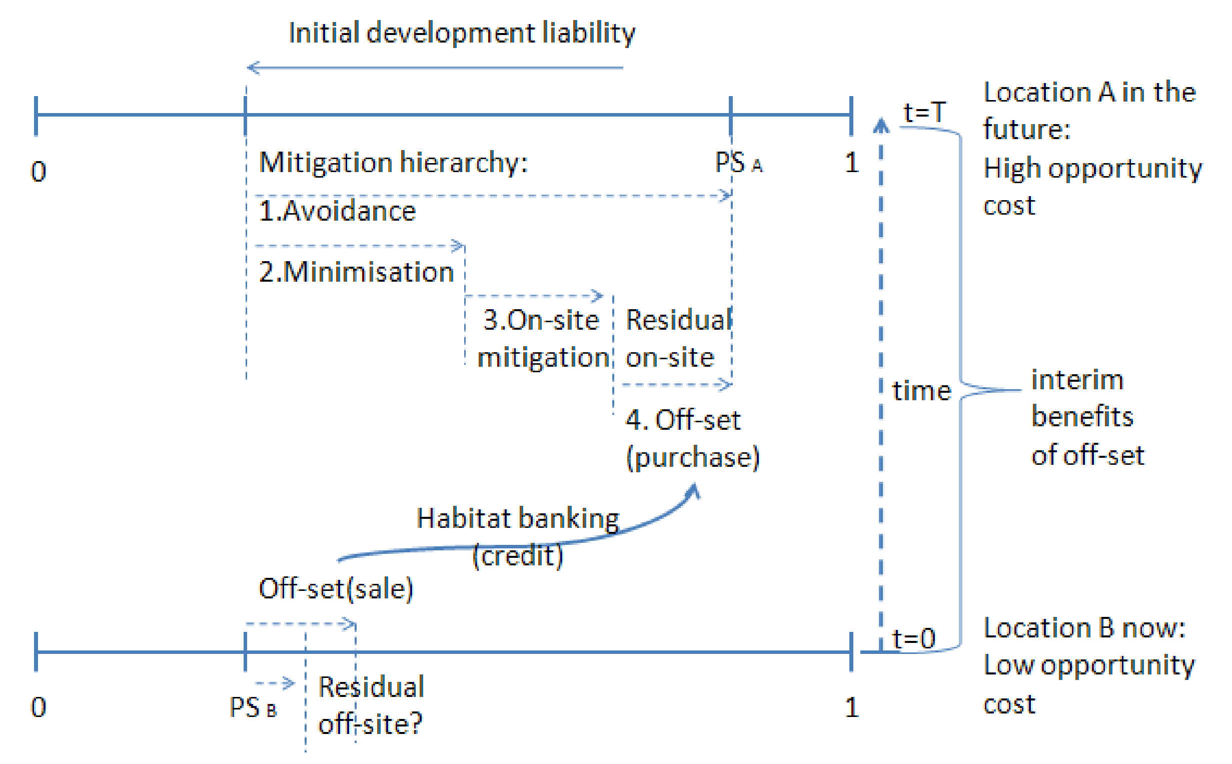

Figure A3 Interim habitat loss in habitat banking (no ecosystem dynamics)

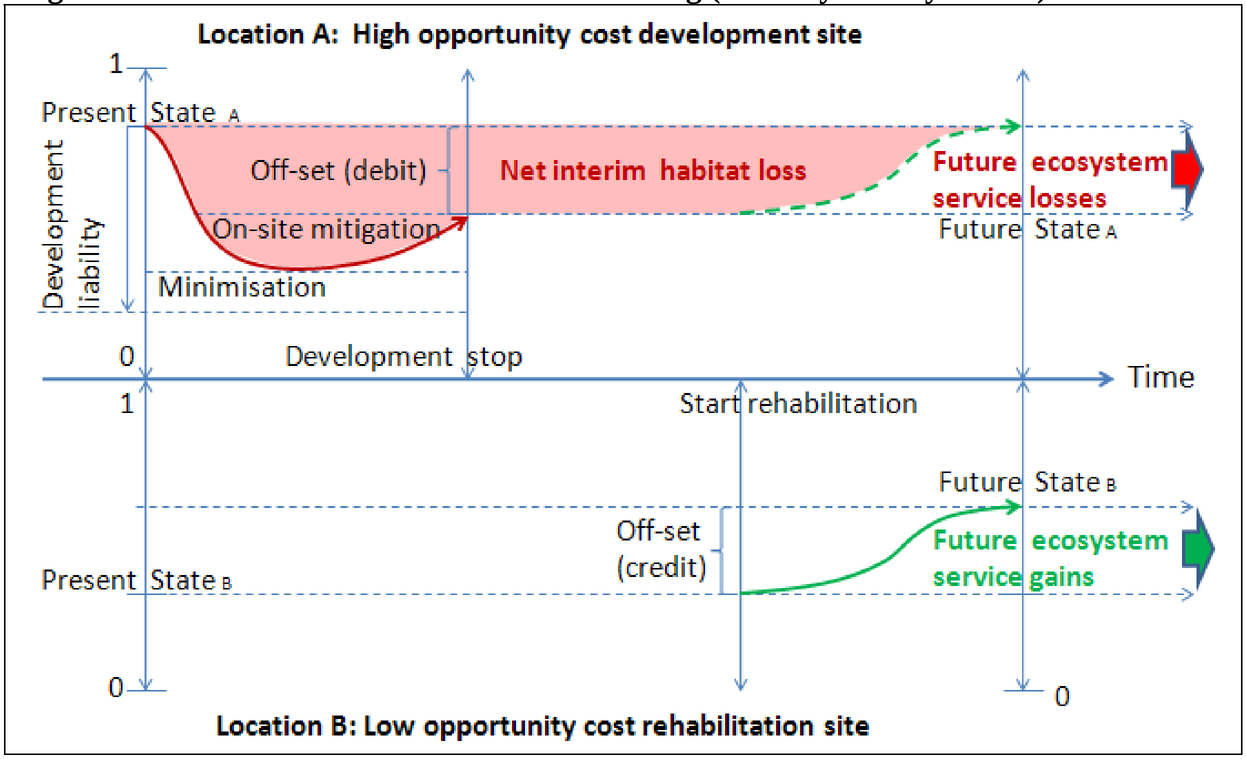

Figure A4. Equivalence of ecosystem services gains and losses with ecosystem dynamics

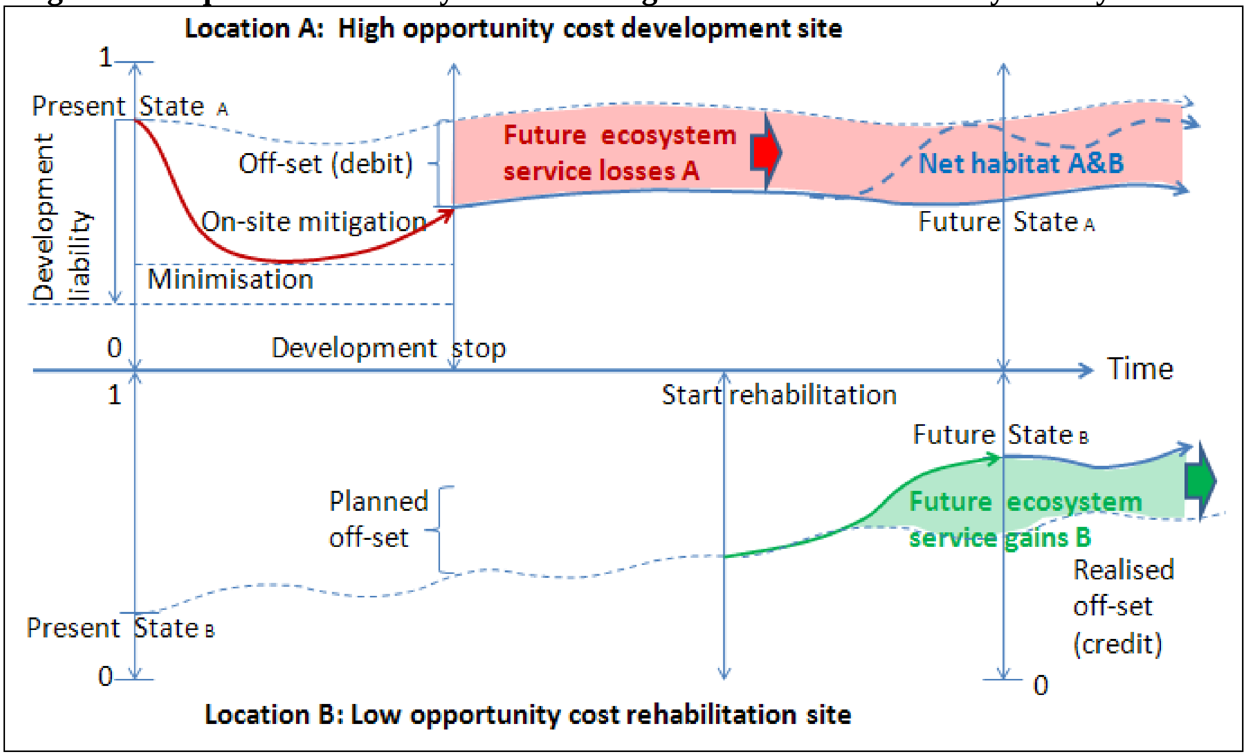

Table A1 Restoration time-scales for selected habitat types

Table 4.1. The feasibility of restoring selected habitat types and their relative time-scales

Habitat	Time-scale	Notes
Temporary pools	1-5 years	May never support some faunas e.g. *Triops* and *Cheirocephalus*, but rapidly colonised by water beetles.
Eutrophic ponds	1-5 years	Creatable provided adequate water supply. Readily colonised by water beetles and dragonflies but faunas restricted to those with limited specialisms. Include ponds created for Great Crested Newts *Triturus vulgatus*.
Mudflats	1-10 years	Dependent upon position in tidal frame and sediment supply.
Eutrophic grasslands	1-20 years	Dependent upon availability of propagules.
Reedbeds	10-100 years	Will readily develop under appropriate water conditions.
Saltmarshes	10-100 years	Dependent upon availability of propagules, position in tidal frame and sediment supply.
Oligotrophic grasslands	20-100 years +	Dependent upon availability of propagules and limitation of nutrient input.
Chalk grasslands	50-100 years +	Dependent upon availability of propagules and limitation of nutrient input.
Yellow Dunes	50-100 years +	Dependent upon sediment supply and availability of propagules. More likely to be restored than re-created.
Heathlands	50-100 years +	Dependent upon nutrient loading, soil structure and availability of propagules. No certainty that vertebrate and invertebrate assemblages will arrive without assistance. More likely to be restored than re-created.
Grey dunes and dune slacks	100-500 years	Probably not recreatable but potentially restorable.
Ancient Woodlands	500 - 2000 years	No certainty of success if ecosystem function is sought - dependent upon soil chemistry and mycology plus availability of propagules. Restoration a possibility for plant assemblages but questionable for rarer invertebrates.
Vegetated shingle structures	500 - 5000 years	Dependent upon sediment supply and coastal processes. Essentially un-recreatable.
Blanket Bogs	1,000 - 5,000 years	Probably un-recreatable but will form in these timescales.
Raised Bogs	1,000 - 5,000 years	Probably un-recreatable but will form in these timescales.
Limestone Pavements	10,000 years	Un-recreatable but will form if a glaciation occurs.
Pingoes	10,000 years	Un-recreatable but will form if a glaciation occurs.
Turloughs	10,000 years	Un-recreatable but will form if a glaciation occurs.

Source: Morris and Barham, 2007

Source: cited in EFTEC (2010:80)

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